



K. O. I
11/14/95 86767
000011

46050 Manekin Plaza • Suite 100 • Sterling, Virginia • 20166

703 • 444 • 7000 FAX: 703 • 444 • 1685

November 16, 1995

VIA EXPRESS MAIL

Mr. Thomas Hilbert
Environmental Engineer
Winnebago Reclamation
8403 Lindenwood Rd.
Rockford, IL 61109

Subject: Corrections to the Pagel Landfill Application for "Significant
Modification" of the existing Permit, Log No. 1995-250
GeoTrans Project No. 7740-007

Dear Tom,

Enclosed please find the formal response to those items in the application that require correction or additional information to satisfy IEPA comments. The response addresses the comments specified in the IEPA letter dated October 6, 1995 and discussed during the meeting on October 24, 1995. To this end, four reports have been modified: (1) Groundwater Management Zone Application; (2) Groundwater Impact Assessment; (3) Groundwater Monitoring Plan; and (4) Groundwater Remedial Alternative Analysis and Preliminary Design (Air Sparging). The latter has been revised to include an evaluation of remedial alternatives and is now entitled "Corrective Action Measures Assessment and Preliminary Design."

The following replacement pages should be inserted into each respective document to address the comments or reflect changes in pagination.

Groundwater Management Zone Application:

1. Replace Table of Contents pages ii and iv,
2. Replace pages 10 through 14 and 28, and
3. Insert Attachment 1 and associated Figures A1 through A6 at page 32.

Groundwater Impact Assessment:

1. Replace Table of Contents pages ii and v,
2. Replace text pages 24, 30, 31, 90, 91, 92, 93, 95, and 96, and
3. Insert Attachment 1 (pages 97-99).

Groundwater Monitoring Plan:

1. Replace Table of Contents pages ii and v and
2. Replace pages 1, 2, 13, 14 (Figure 3.1), 15, 16, 17 (Figure 3.2), 23, 24, 26, 33-36, and 40-59.

Groundwater Remedial Alternative Analysis and Preliminary Design (Air Sparging):

1. This document has been revised for IEPA and is entitled "Corrective Action Measures Assessment and Preliminary Design."

If you have any questions or need clarification, please do not hesitate to contact me at (703) 444-7000.

Sincerely,



Alex Vincent
Senior Hydrogeologist

av/AV

enclosures: as stated

cc: D. Feezor (Andrews) for C. Liebman, IEPA (w/encl. - 4 copies)
B. Schorle, USEPA Region 5 (w/encl. - 2 copies)
R. Rajaram, PRC Environmental (w/encl. - 1 copy)
D. Burnell, GeoTrans (w/encl. - 1 copy)
P. Rich, GeoTrans (w/encl. - 1 copy)

**CORRECTIVE ACTION MEASURES ASSESSMENT
AND PRELIMINARY DESIGN**

Prepared for:

Winnebago Reclamation
8403 Lindenwood Road
Rockford, Illinois 61109

Prepared by:

GeoTrans, Inc.
46050 Manekin Plaza, Suite 100
Sterling, Virginia 20166



Peter A. Rich, P.E.
Illinois License #062-050190

GeoTrans Project No. 7740-007

November 16, 1995

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1 INTRODUCTION

This report provides an evaluation of various remedial alternatives (corrective action measures assessment) and presents the preliminary design of a selected remedy for impacted groundwater at the Winnebago Reclamation Landfill (WRL) Site in Winnebago County, Illinois. This report is necessary due to the findings presented in a previous report entitled "Groundwater Remedial Alternative Analysis and Preliminary Design" (GeoTrans, 1995c). The following report describes and evaluates several alternatives and includes the preliminary design of the remedy selected based on the time required to complete the remedy, effectiveness, performance, reliability, and ease of implementation. The evaluation is submitted in accordance with Title 35 Illinois Administrative Code (IAC) 811.324(e) which requires that the results of a corrective action measures assessment be presented to the Illinois EPA.

This report is organized into six sections, including this introduction. Characterization of water quality parameters which assist in the determination of the selection and design of remedial processes is included as Section 2. This section is based on the data compiled for this report and others (GeoTrans, 1995a,b,c,d). *Ex-situ* and *in-situ* treatment processes are discussed and evaluated in Section 3 based on guidance set forth in Title 35 Illinois Administrative Code (IAC) 811.324(d). Table 3.2-1 is presented to compare the alternatives in accordance with 35 IAC 811.324(d) regulations. Section 3 also demonstrates compliance of the selected remedy in accordance with the 35 IAC 811.325 guidelines. The recommended treatment alternative is presented in Section 4. An analysis of the costs for the implementation of the recommended remedy and schedule are also included in Section 4. Monitoring and operating considerations in accordance with 35 IAC 811.326 are included in Section 5 and 6, respectively.

1.1 PURPOSE AND BASIS OF REMEDIATION

The purpose of groundwater remediation in conjunction with source removal actions, including landfill capping and leachate collection at the WRL site is to:

1. reduce concentrations of leachate-derived constituents in groundwater to applicable groundwater quality standards (AGQS); and
2. to prevent offsite migration of inorganic, volatile, and semi-volatile organic compounds in the groundwater associated with the site that are in excess of applicable or relevant and appropriate requirements (ARARs).

The basis for the requirements of remediation are provided by Illinois regulations and the Record of Decision (ROD) issued on June 28, 1991. The ROD states that groundwater is to be extracted in such a manner as to provide containment of groundwater contaminants to prevent their offsite migration. Extracted groundwater will be treated to standards set forth by ARARs. These ARARs include, but are not limited to, regulations set forth by the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), Superfund Amendments and Reauthorization Act of 1986 (SARA), the National Oil and Hazardous Substances Pollution Contingency Plan (NCP), and the State of Illinois. Remedial actions will also address contamination which may be present as a result of the downgradient migration of groundwater associated with offsite sources of contamination.

1.2 REMEDY BACKGROUND

The preliminary design in a previous report (GeoTrans, 1995c) included groundwater recovery and treatment by air stripping as required in the ROD. Since the effect of elevated concentrations of ammonia in groundwater on air stripping treatment requirements was not considered in reports prior to the ROD, operating and maintenance costs were considerably underestimated in the ROD. Due to the relatively high concentration of ammonia in the groundwater, remedial alternatives other than groundwater recovery and air stripping are, therefore, likely to be more efficient and cost effective. Although the ROD specifies

groundwater extraction to achieve the goal of preventing offsite migration of groundwater contaminants, innovative *in-situ* technologies are being tested and implemented at similar sites to accomplish these same goals. In a meeting on May 19, 1995 USEPA and Winnebago Reclamation Service, agreed to consider air sparging and natural attenuation technologies. Air sparging and natural attenuation have been successfully utilized at similar sites. The WRL site hydrogeology and low levels of volatile organic compounds (VOCs) and inorganic constituents are well suited for these remedial measures. Due to this suitability and the difficulty associated with treating and discharging recovered groundwater at the site, the ROD wording requiring groundwater extraction as the remedy should be amended.

2 GROUNDWATER TREATMENT CHARACTERIZATION

Characterization of water quality parameters which assist in the determination of the selection and design of remedial processes is presented below. This section also discusses the flowrate and chemical parameters which characterize the groundwater influent to an *ex-situ* treatment system from a network of extraction wells. The *ex-situ* system influent concentrations represent the initial groundwater concentrations that an *in-situ* system must address. The advantage of *in-situ* treatment is briefly discussed. In addition, the expected quality of water to be discharged from an *ex-situ* system effluent is discussed. Identification and determination of these parameters is an important aspect of the selection and design of the corrective action. The parameters were estimated based on observed groundwater concentrations along the western part of the landfill.

2.1 EXPECTED INFLUENT/INITIAL GROUNDWATER CHARACTERISTICS

The groundwater extracted by a recovery well network will compose the influent to an *ex-situ* treatment system. Determination of the quantity of water for an *ex-situ* system was examined using a calibrated groundwater flow model and was discussed in GeoTrans, 1995c. The model estimated that three wells pumping at a cumulative rate of approximately 450 gpm will be sufficient to contain the regions of contaminated groundwater at the site. The placement of the *in-situ* system will be in the same location as the recovery well network; therefore, as stated above, the initial concentrations are expected to be similar for the *in-situ* system.

The expected concentration of chemical constituents in the influent were determined by analysis of recent and historical groundwater samples from representative wells along the western edge of the site in the general area of the proposed extraction wells. The wells were selected based on screened intervals in the upper and lower zones of the sand and gravel aquifer. These wells included RW01, P3R, P4R, G104, P1, and MW106. The sources of chemical constituent information include results of analytical tests performed on samples

taken from these wells and from water produced during aquifer testing procedures at the recovery wells. The data are located in a comprehensive water quality database constructed by GeoTrans from historical site data. The results show elevated concentrations of ammonia nitrogen ($\text{NH}_3 - \text{N}$) and several VOCs. Ammonia nitrogen and several VOCs are present at levels that require treatment prior to surface water discharge. Total Suspended Solids (TSS) and Total Dissolved Solids (TDS) were also found at elevated levels; however, both are expected to decrease during pumping from the system recovery wells. In addition, elevated concentrations of nitrate and nitrite were detected upgradient and at the site. The presence of nitrate is probably caused by both transformation of ammonia and upflow from high nitrate concentrations in bedrock. Elevated nitrate concentrations are observed in both unconsolidated and bedrock sediments upgradient of the site, and are most likely caused by offsite agricultural activities. Any landfill-related nitrate concentrations are expected to decline as ammonia is withdrawn and removed, thus limiting the transformation of ammonia to nitrate and nitrite. Design consideration was also given to constant pumping over time, which will draw clean groundwater into the contaminated regions, and along with source removal actions, will lessen influent concentrations of all contaminants. This action will decrease the concentrations of contaminants until a steady state is approached. The phenomenon is expected to result in influent concentrations of zinc and chloride within discharge limits. Hardness and alkalinity do not have associated discharge level requirements, although the relatively high concentrations present will affect the *ex-situ* treatment system design. Expected influent concentrations are summarized in Table 2.1-1. The design of the *ex-situ* treatment system will be governed, in part, by these influent concentrations.

An *in-situ* treatment system would be required to treat groundwater from the initial conditions to prevent migration of contaminants at levels of concern from the site. Based on the ROD and an evaluation of groundwater data, these contaminants consist of low level VOCs which are amenable to *in-situ* air sparging. All of the VOCs and moderate amounts of ammonia are expected to be removed through volatilization. It should be noted that treatment of inorganic compounds is not addressed by the ROD selected remedy. However, *in-*

Table 2.1-1. Expected influent and effluent concentrations for *ex-situ* groundwater remediation alternatives at Winnebago Reclamation Landfill site.

Constituent	Expected Influent Concentration	Expected Effluent Standard	Source of Effluent Standard
Alkalinity	746 mg/L	N/A ¹	
Ammonia Nitrogen	75-125 mg/L	1.5 mg/L (summer) 3.0 mg/L (winter)	IEPA ²
Arsenic	9.6 µg/L	190 µg/L	35 IAC 302.208(d)
Barium	604 µg/L	5 mg/L	35 IAC 302.208(e)
Benzene ³	<5 µg/L	5 µg/L	Federal MCL
Biological Oxygen Demand (BOD)	12 mg/L	10-30 mg/L	35 IAC 304.120(a) / IEPA
Bromomethane	<10 µg/L	N/A	
2-Butanone ³	<100 µg/L	N/A	
Calcium	63 mg/L	N/A	
Carbon Tetrachloride ³	<5 µg/L	5 µg/L	Federal MCL
Chemical Oxygen Demand (COD)	102 mg/L	N/A	
Chloride	103 mg/L	500 mg/L	35 IAC 302.208(e)
Chlorobenzene ³	<5 µg/L	N/A	
Chloroethane ³	<10 µg/L	N/A	
Chloromethane ³	<10 µg/L	N/A	
1,4 Dichlorobenzene ³	<10 µg/L	75 µg/L	Federal MCL
1,1 Dichloroethane	<7 µg/L	N/A	
1,2 Dichloroethane ³	<5 µg/L	5 µg/L	Federal MCL
1,1 Dichloroethene ³	<5 µg/L	7 µg/L	Federal MCL
1,2 Dichloroethene (cis) ³	<5 µg/L	70 µg/L	Federal MCL
1,2 Dichloroethene (trans) ³	<5 µg/L	100 µg/L	Federal MCL
Dichloromethane ³	<5 µg/L	5 µg/L	Federal MCL
1,2 Dichloropropane	<5 µg/L	5 µg/L	Federal MCL
Ethylbenzene	<5 µg/L	700 µg/L	Federal MCL
Hardness	212-481 mg/L	N/A	
Iron	1.3-4.1 mg/L	1 mg/L	35 IAC 302.208(e)

Table 2.1-1. Expected influent and effluent concentrations for *ex-situ* groundwater remediation alternatives at Winnebago Reclamation Landfill site (continued).

Constituent	Expected Influent Concentration	Expected Effluent Standard	Source of Effluent Standard
Magnesium	49 mg/L	N/A	
Manganese	0.34 mg/L	1 mg/L	35 IAC 302.208(e)
Nickel	0.16 mg/L	1 mg/L	35 IAC 302.208(e)
Nitrite + Nitrate Nitrogen	1.4-4 mg/L	10 mg/L	Federal MCL
Phenol	2-12 µg/L	100 µg/L	35 IAC 302.208(e)
Potassium	86 mg/L	N/A	
Sodium	147 mg/L	N/A	
Sulfate	107 mg/L	500 mg/L	35 IAC 302.208(e)
Toluene	<5 µg/L	1000 µg/L	Federal MCL
Total Dissolved Solids (TDS)	1000 mg/L	1000 mg/L	35 IAC 302.208(e)
Total Organic Carbon (TOC)	30 mg/L	N/A	
Total Suspended Solids (TSS)	<423 mg/L	12-15 mg/L	35 IAC 304.124(a) / IEPA
Tetrachloroethene	2-14 µg/L	5 µg/L	Federal MCL
1,1,1 Trichloroethane	4 µg/L	200 µg/L	Federal MCL
Trichloroethene	1-6 µg/L	5 µg/L	Federal MCL
Vinyl Chloride	<10 µg/L	2 µg/L	Federal MCL
Xylenes (total)	<2 µg/L	1000 µg/L	Federal MCL
Zinc	<1.5 mg/L	1 mg/L	35 IAC 302.208(e)

¹N/A: Not Applicable. No general use water quality standard, general effluent standard, or MCL presently exists.

²IEPA: Standard was derived after consultation with the IEPA Water Quality Planning Division.

³Expected influent concentrations less than the Practical Quantitation Limit (PQL)

Note: Bold parameters indicate those considered in system design.

situ air sparging would have an advantageous affect by adding dissolved oxygen (DO) to the groundwater, enhancing biodegradation processes such as nitrification, and eliminating reducing conditions. The proposed limiting of leachate source material by capping and pumping will allow aerobic degradation including nitrification of a lesser mass of ammonia to take place immediately downgradient of the landfill. As the source of the plume is nearly eliminated the outlying elevated levels of landfill constituents will quickly undergo natural biodegradation or attenuation.

2.2 EXPECTED *EX-SITU* TREATMENT SYSTEM EFFLUENT CHARACTERISTICS

The effluent from the *ex-situ* treatment system is proposed to be discharged to a local surface water body, Kilbuck Creek, which is adjacent to the site. Alternatively, discharge to the Publicly Owned Treatment Works (POTW) may be cost effective if hydraulic capacity is available and fees are approximately \$2 per 1000 gallons or less. Reinjection permitting has not been examined but may prove a viable option if ammonia discharge standards are comparatively relaxed.

Data show a range of creek flow rates from 16 cubic feet per second (cfs) to 188 cfs in 1988 (USGS, 1988). Assuming that these flow rates are representative, then discharge of as much as 1.6 cfs (700 gpm) from the *ex-situ* treatment system is expected to have relatively insignificant effects on the overall flow rate of the creek. The stream should provide significant dilution resulting in only a slight increase in ambient concentrations. The demonstration of dilution by the stream or the knowledge of background concentrations is not likely to affect the discharge levels specified in future permits. Discharge levels are generally determined with respect to the preservation of local aquatic flora and fauna.

Discharge standards specified in National Pollutant Discharge Elimination System (NPDES) permits issued by the Illinois Environmental Protection Agency (IEPA) will govern, in part, the selection and design of the *ex-situ* treatment processes. Expected NPDES discharge standards were derived after examining the relevant sections of Title 35 of the

Illinois Administrative Code IAC, the Code of Federal Regulations (Title 40), and communicating with representatives of the IEPA. Details of records were presented in GeoTrans, 1995c. Expected discharge standards pertaining to constituents detected at the site are listed in Table 2.1-1, with influent constituents close to or exceeding discharge standards, or of importance to the design, in bold. The discharge standards will be used in conjunction with the estimated influent concentrations to recommend an *ex-situ* groundwater treatment system for use at the WRL Site.

3 GROUNDWATER REMEDIATION SYSTEM DESCRIPTION AND EVALUATION

This section includes the evaluation of potential remediation technologies applicable to contaminated groundwater at the WRL site which could be employed in conjunction with the planned source removal actions. Brief descriptions and projected efficiencies are presented in this chapter. One *in-situ* and four *ex-situ* groundwater treatment options are included. The *ex-situ* options consist of one physical and three biological treatment alternatives for extracted groundwater. Additional polishing steps would be required, in some cases, to reach the required NPDES discharge standards.

Detailed design and evaluation of the originally selected alternative of air stripping, and *ex-situ* technology, was included in the report *Groundwater Remedial Alternative Analysis and Preliminary Design* (GeoTrans, 1995c). The final alternative presented in this section is *in-situ* air sparging, which addresses VOCs and moderate amounts of ammonia by direct removal of these and other contaminants through physical and biological processes. This technology parallels the *ex-situ* air stripping technology selected in the ROD with a similar *in-situ* technology which utilizes the same physical volatilization process of air sparging and additionally increases dissolved oxygen levels to provide enhanced biodegradation of constituents.

3.1 INITIAL IDENTIFICATION OF TECHNOLOGIES AND PROCESS OPTIONS

In order to address the VOC and ammonia present in the groundwater, five potentially applicable technologies are identified in this section. *Ex-situ* biological treatment process options presented include fluidized bed reactor, sequencing batch reactor, and constructed wetlands treatment. Air stripping, which is an *ex-situ* physical treatment alternative, is presented in this section. Air sparging, is included in this section as an *in-situ* treatment option.

3.1.1 EX-SITU TREATMENT – AIR STRIPPING

Air stripping is the process in which dissolved gases and volatile compounds in water are removed from solution by liquid to air mass transfer. Typical strippers are configured similarly to industrial cooling towers using a counter-current flow scheme. Water is allowed to trickle downward over a contact media of plates, rings, spheres, or corrugated surfaces while clean air is forced upward through the tower and contact media by blowers or fans. The mass transfer of the volatiles and dissolved gases to air is governed by Henry's Law, which is temperature sensitive, as temperature has an affect on the vapor pressures of dissolved constituents (McGhee, 1991). As a result, temperature is an important design parameter for ammonia stripping. The fraction of ammonia present as NH_3 at a given pH increases with temperature. Efficiency is greatly improved at higher temperatures. In addition, pH also has an affect on the ionization of ammonia in solution. Optimal pH for ammonia stripping is in the range of 10.8-11.5 with 11.0 as a common design parameter (Tchobanoglous et al., 1987). Theoretical gas (air) requirements can be determined from the principle of mass conservation with design requirements typically 1.5 to two times greater than this value (Reynolds, 1982). Tower heights range from 15 to 35 feet (4.6 to 10.7 meters) and diameters are dependent on air flow requirements. Tower porosities are usually 95 percent with a surface area to volume ratio of 80-100 m^2/m^3 (24.4-30.5 ft^2/ft^3) (Tchobanoglous et al., 1987). Hydraulic surface loading rates are typically 1-2 gpm/ft^2 (58.8-117.6 $\text{m}^3/\text{m}^2/\text{day}$) (Reynolds, 1982).

Air stripping offers the most cost-effective removal process for most VOCs under high flow rate conditions (McGhee, 1991). In addition, it has proven effective for ammonia removal at many advanced wastewater treatment plants as well as landfills and hazardous waste sites. As a result, air stripping for the removal of VOCs and ammonia is applicable for use at the WRL Site. The process will require pH adjustment, and the removal efficiency may be optimized by heating of extracted water. Scaling of the contact media will occur due to the pH adjustment and the hardness of the water. Frequent washing and scale removal will be required. Overall removal efficiency of a single stripper unit may be as high as 95 percent

(Metcalf and Eddy, 1985). Air stripping towers in series may be used to increase efficiency as necessary.

Several treatment processes are required prior to air stripping. These processes include alkali pH adjustment and potential clarification, magnetic treatment, and temperature elevation. In order to increase the mass transfer efficiency of ammonia from groundwater to air, pH adjustment processes should be designed in the system. The pH value of the groundwater should be increased from seven to 12 before going through the air stripping units and then decreased to 8.5 before being discharged to the creek. An incline plate clarifier will be used, if necessary, for the removal of total suspended solids and other inorganic precipitates after pH adjustment to 12. In addition to clarification, magnetohydrodynamic (MHD) treatment is another process which minimizes scale buildup in piping and subsequent fouling of the air stripping towers. The dynamic nature of this stripping operation is a result of the physical and chemical qualities of ammonia and the groundwater. One of the primary parameters driving the stripper design is the average temperature of groundwater. A potential action to further increase removal efficiency is to preheat the influent to the air strippers. Heat generated from the WRL gas extraction facility/sludge drying facility could be diverted to heat the water to a more desirable temperature.

The removal of ammonia from the extracted groundwater can be achieved by air stripping. The towers will operate in a countercurrent scheme with air flowing upward and water flowing downward entering through distributors at the top of the tower. Mist eliminators installed at the top of the tower will prevent the loss of water through the exhaust vent. Increased air/water contact will be provided by packing material located throughout the length of the tower. Air carrying the stripped ammonia and minimal VOCs will be discharged to the atmosphere through the exhaust vent at the top of the tower. Water will be collected at the bottom of the tower and then transferred to other processes by transfer pumps.

Four sets of two towers operating in series will be required for a removal efficiency of 98 percent or greater at an estimated flow of 450 gpm. Dual transfer pumps rated at 500 gpm

each will be provided as the four transfer pumps indicated in the process flow diagram and will allow flexibility for potential recycling of treated water to dilute the influent stream. The effluent from the system will contain no more than 1.5 mg/L $\text{NH}_3\text{-N}$ between May and September and no more than 3 mg/L between October and April. The seasonal discharge standard of 3 mg/L between October and April may be repealed based on a proposed rule of the Illinois Pollution Control Board so that the year round standard would be 1.5 mg/L. VOCs will be effectively stripped to below MCL standards, as ammonia stripping is many times more intensive. The towers will be 12 feet in diameter and 45 to 50 feet in height with a packing height of 30 to 35 feet. These dimensions will support the required surface loading rate of one to two gpm/ft². The packing material will consist of PVC or PE plates or trays in a shape similar to a sinusoidal wave. The required air flow to each tower is 115,000 cubic feet per minute (cfm) and will be supplied by a 50 hp blower. This air flow rate will meet the recommended minimum air-to-water ratio of 1000:1. The treated water will exit the system at a pH too high for direct discharge to surface water. Therefore, pH adjustment by HCl acid is necessary.

Estimates of the proposed pump-and-treat system capital and annual operation and maintenance costs are \$3,109,245 and \$780,428, respectively. Additional contingencies for total construction cost and annual operation and maintenance are \$350,000 and \$248,280, respectively. The costs are elevated above those estimated in the ROD due to the requirement of ammonia removal. Air stripping of ammonia requires pH adjustment, approximately 20 times the stripping tower surface and 50 times the air flow rate as compared to standard VOC air stripping. Foundations for the stripper and a significant treatment system building are required for chemical feed and water transfer.

The operating requirements include significant power to run motors especially for air stripper blowers, designated system operators, and regular system maintenance. Costs may be reduced as influent concentrations decrease and individual air strippers can be taken off line, and possibly by the addition of magnetic sequestering to reduce scaling. However, if solids removal is required to reduce scaling, both capital and operating costs will be increased.

3.1.2 EX-SITU TREATMENT – TRICKLING FILTER/FLUIDIZED BED REACTOR

Trickling filters were discussed in detail in the previous report to Winnebago Reclamation Service (GeoTrans, 1995c). In summary, trickling filters are fixed film aerobic systems in which wastewater flows by gravity through a contact media of rock or synthetic materials. Attached microbial growth oxidizes soluble organics and nitrogen present in the wastewater. Multi-stage filter systems or filters with deep bed depths and low loadings can sustain nitrification. Denitrification may be achieved in similar filter configurations, however, application of this technology is rare, as its biological and physical mechanisms are poorly understood and difficult to control and optimize. As with other conventional technologies, a supplemental carbon source, such as methanol, is needed for denitrification to proceed. Successful denitrification is typically conducted in suspended growth systems (Metcalf and Eddy, 1985).

Consultation with a vendor has indicated a potentially applicable trickling filter design for use at the WRL site. The system is designed to provide a 95 percent maximum removal of ammonia nitrogen with a two stage trickling filter. Each filter will be 52 feet in diameter and have a media depth of 20 feet. The packing medium will be honeycombed circular plastic (Bio-Pac SF #30 or equivalent). A recycle rate of 50 percent is required to ensure proper oxygen transfer to the biomass. In addition, high alkalinity (680 mg/L) and a consistent phosphorus concentration (1 mg/L) of the wastewater must be maintained. It is unlikely that solids filtration prior to the trickling filters will be necessary, but sloughing of the biomass in the filter units will require the use of a secondary clarifier. The clarifier should be sized to 42 feet in diameter to provide a surface loading rate of 450 gal/ft². Capital costs of this system are projected at \$530,000. Operation and maintenance (O&M) costs would be greater than \$300,000 per year.

The proposed design will not provide a sufficient level of treatment to meet expected NPDES discharge standards. A polishing operation with 70 percent removal efficiency is necessary to remove remaining ammonia. It is important to note that this trickling filter system is designed to provide nitrification only and subsequent denitrification will require additional treatment operations. Potentially applicable technologies include wetlands

treatment, air stripping, and conventional suspended growth systems. A description of air stripping as a polishing step is included in Section 3.2. Wetlands treatment is described in Section 3.3. Optimal operation of the trickling filter system will occur in temperatures greater than 7° C (49° F), and decreased efficiency is to be expected during the cooler temperatures.

Fluidized-bed reactors (FBRs) are fixed film systems in which the solid, bedding material or packing media is colonized by microorganisms while being suspended by water flowing upward through the tank. The packing material can be inert (sand, plastic, or coal) or active (granular activated carbon). Aerobic and anaerobic FBRs have seen prior use at sites. For aerobic systems, air is diffused from the bottom of the bed to supply the necessary amount of oxygen. FBRs have some advantages over traditional packed bed systems such as trickling filters. Smaller bed particles can be used because gas bubbles can pass through the bed more easily. The use of smaller particles yields a higher biofilm surface area which allows the system to handle increased organic loadings. The bedding of FBRs expands over time and does not clog as in trickling filters. Growth can be controlled by removing particles from the top of the bed, clarifying the biomass, and returning the particles to the bottom of the reactor (Noyes, 1994).

FBRs which utilize granular activated carbon (GAC) as the bed material are referred to as biological-activated carbon systems or FBR-GAC systems. The two main removal mechanisms in this system are biodegradation and adsorption, which occur simultaneously. The GAC protects the system from organic shock loads, increases retention time of less readily biodegraded organics, and adsorbs refractory organic compounds. Microorganisms in the reactor affix themselves to the GAC surface and have shown to regenerate inactive sorption sites on the carbon. The main disadvantage of this type of system is the large capital investment requirements in GAC. As with other FBR systems, little information is known on vapor emissions from the reactor. Vapor emissions from diffused air systems may be significant depending on the influent characteristics (Noyes, 1994).

Consultation with a vendor of this technology has produced a conservative design for a FBR-GAC system applicable to the WRL site. The proposed system would provide

sufficient ammonia and VOC removal at acceptable costs. The system consists of a two stage diffused air system and is designed to handle 450 gpm and provide a hydraulic residence time of 135 minutes. Each reactor would be 18 feet in diameter with a bed depth of 16 feet and would initially require 61000 pounds of carbon. Carbon would be replaced at a rate of 17 pounds per day. In addition, a nutrient slurry must be fed to the system at a rate of 0.5 pounds per day. Other major operational costs will be incurred by the supply of oxygen (2700 lb/d) in either a pure form or from air. The system is expected to have capital costs ranging from \$1,550,000 to \$1,700,000 and annual O&M costs of \$85,000 to \$102,000. Vapor emissions controls are not likely required as the influent to the system will contain low concentrations of VOCs.

3.1.3 EX-SITU TREATMENT – SEQUENCING BATCH REACTOR

A sequenced batch reactor (SBR) is one of the three main forms of suspended growth systems of waste activated sludge (WAS) processes. Single stage SBRs were the most common form of WAS processes for domestic sewage treatment prior to 1950. They have seen a resurgence in the last 15 to 20 years, especially in the hazardous waste remediation industry (Tchobanoglous et al., 1987). Full scale SBR operations have shown to be capable of achieving the effective removal of biological oxygen demand (BOD), suspended solids (SS), nitrogen, and phosphorus and the treatment of some hazardous wastes (Irvine et al., 1985).

The SBR may simply be described as a periodic, time-oriented activated sludge process. The normal sequencing steps carried out are: (1) fill; (2) react; (3) settle (sedimentation and clarification); (4) draw (or decant); and (5) idle. SBRs can operate in a continuous flow scheme as a series or may operate in parallel (Irvine et al., 1985).

The conversion of inorganic nitrogenous compounds to nitrogen gas occurs in SBRs in the same manner as in conventional nominal plug flow (PF) and continuous flow stirred tank reactors (CFSTR). The conversion of ammonia to nitrate (nitrification) is carried out by autotrophic organisms which require dissolved oxygen and little to no organic carbon. Nitrate to nitrogen reactions (denitrification) are carried out by heterotrophic microbes in

anoxic conditions. These heterotrophs require a carbon source for growth and energy before utilizing nitrate as an electron acceptor and converting it to nitrogen gas. Systems with low BOD loading will require the addition of a supplemental carbon source, typically methanol (Reynolds, 1982). Nitrification occurs readily in low loaded SBR systems, but denitrification is difficult to maintain. Full-scale SBRs have achieved 90 to 95 percent net nitrogen removal, demonstrating that denitrification can be achieved. Removal efficiencies in excess of these amounts have not been practically established (Palis and Irvine, 1985). The remaining five percent to ten percent of nitrogen is in the nitrate form with a very small fraction bound up in organic compounds.

The application of SBR technology to the treatment of groundwater at the WRL site would have limited success. The required ammonia removal efficiency exceeds 98 percent, and vendors of SBR technology will not guarantee nitrogen removal efficiencies greater than 95 percent without major design modifications and increased costs. SBRs, like other conventional WAS processes, generate a sludge of approximately one percent solids. Sludge must be dewatered and disposed. This technology also has significant chemical requirements for the biological processes and sludge stabilization. Power requirements for aeration blowers, mechanical stirring, and transfer processes are also significant. Full time operator(s), a control panel, and treatment system housing are necessary for efficient operation. The cost for an SBR system at the WRL site was estimated from consultation with a vendor of this technology. Capital and start up costs for 95 percent effective ammonia removal at 450 gpm were \$1,305,000 with an annual O&M cost of \$387,000. Removal of the remaining ammonia would have to be achieved by another process such as air stripping.

A treatment scheme potentially applicable to the WRL site would include ammonia removal by a multi-stage system of two parallel operating SBRs and three air stripping towers for the removal of the remaining ammonia and volatile organics. The SBRs would provide nitrification, denitrification, and solids clarification. The two SBRs (34' h x 52' dia., each) would operate on four- to six-hour cycles, where one fills while the other is in the reaction, settling, and decanting stages. Nitrification would require an air supply from external blowers and internal mechanical mixing. Denitrification occurs in an anoxic

environment in which no air is supplied and the substrate is mixed mechanically. Optimum pH for both of these processes is around 7.5, thus it is unlikely that pH adjustment will be necessary. The addition of methanol during denitrification is necessary to supply a supplemental carbon source for microbial growth. Clarification also occurs in the same reactor and produces a sludge of 0.8 percent to 1 percent solids. Waste sludge flow will be optimized shortly after the system is brought on line. Waste sludge will require stabilization by the addition of a polymer and will be dewatered by a belt filter press (BFP) to result in a sludge of 18 percent to 20 percent solids. The dewatered sludge is expected to pass the TCLP test and may be disposed at the site landfill. The BFP will be housed in an equipment building (23' x 20') along with a control panel and space for an analytical laboratory.

The SBR system will achieve 95 percent removal of ammonia and acceptable discharge concentrations of nitrate (<10 mg/L). The remaining ammonia and small fraction of VOCs, if any, will be decreased to discharge standards by air stripping. Three air stripping towers operating in parallel at 150 gpm each will provide 80 percent removal of ammonia. As was discussed in a previous report (GeoTrans, 1995c), a pH of 11 or greater is necessary during air stripping. Therefore, caustic (NaOH) will be added prior to stripping operations. After stripping operations, the pH will be readjusted to acceptable levels for discharge by the addition of hydrochloric acid. The strippers will require the same equipment as mentioned in the previous report, including blowers and a concrete pad. Contrary to the previous report, air strippers in this configuration will not require a recycle flow. A more detailed discussion of ammonia stripping and its design is included in the previous report to Winnebago Reclamation Services (GeoTrans, 1995c). Air strippers for polishing are estimated to have a capital cost of \$670,000 and annual O&M cost of \$360,000.

3.1.4 EX-SITU TREATMENT – CONSTRUCTED WETLANDS TREATMENT

It is generally found that wetlands act as efficient purification systems and nutrient sinks. Permanent water covered or water saturated conditions reduces the net gas exchange between sediments and the atmosphere. As a result, the sediments become mostly anoxic or anaerobic. Organic matter tends to accumulate at the surface of the sediments allowing for a

low bulk density sediment with high water holding capacity and high cation exchange capacity. The detrital sediments and emergent aquatic plant growth, such as macrophytes, provide large surface areas for attached microbial growth (Wetzel, 1975). Wetlands, therefore, have a high potential to accumulate and transform organic material and nutrients.

Natural and constructed wetlands have been successfully used as wastewater treatment systems in North America, Europe, and in developing nations. Current literature stresses the importance of utilizing constructed systems rather than natural systems, as long-term damage to natural wetland ecosystems from wastewater applications has not been fully assessed. Constructed systems for wastewater treatment may, in some locations, provide several advantages compared to conventional and advanced secondary treatment systems. Advantages may include low construction and operation and maintenance costs with low energy and labor requirements. In addition, the systems are typically more flexible and less susceptible to variations in loading rates than conventional systems. The major disadvantage of constructed wetland treatment systems is the large land area required and possible decreased performance during winter in temperate climates (Brix, 1993).

The removal mechanisms in wetlands systems are biological, chemical, and physical. Suspended solids are removed by sedimentation and filtration. Some nutrients and metals are also removed by these processes. Soluble organics and BOD are mostly degraded by aerobic bacteria attached to plant and sediment surfaces. Anaerobic degradation may also be significant, especially in sediments and during oxygen depleting periods in the water column. The oxygen required for aerobic processes is supplied by atmospheric diffusion, photosynthetic production in the water column, leakage from macrophyte roots, and by engineered, mechanical aeration. The major removal mechanism of nitrogen in constructed wetlands is nitrification-denitrification. Ammonia is oxidized to nitrate by nitrifying bacteria in aerobic zones, and nitrates are converted to nitrogen gas (N_2) in anoxic zones. The oxygen required is supplied by the aforementioned processes. Nitrogen may also be taken up by plants and incorporated into their biomass. This uptake is generally less significant than denitrification. Conversion of ammonium to ammonia gas and subsequent volatilization may be significant in systems with open water, under conditions where algal photosynthesis

increases pH levels to above the pK_a value of ammonium (9.26 at 25°C). Phosphorus removal is achieved by adsorption, complexation and precipitation reactions with aluminum, iron, calcium, and clay minerals in the sediment (Brix, 1993).

The majority of constructed wetland treatment systems are macrophyte based and are generalized by several forms:

1. Free floating macrophytes
2. Rooted emergent macrophytes
3. Submerged macrophytes
4. Multi-stage systems

Free floating macrophyte systems mostly involve the use of the common water hyacinth (*eichhornia crassipes*). Nutrient removal is the hyacinth's primary function as it incorporates phosphorus and nitrogen into its biomass. Frequent harvesting of hyacinths is required to maintain maximum productivity as well as to remove the nutrients incorporated in biomass. Induced aeration of this systems aids in effectiveness of soluble organic and nutrient removal by both microbes and macrophytes. Hydraulic residence times vary according to wastewater characteristics, but usually range between five and 15 days. One disadvantage of the hyacinth systems is reduced effectiveness below 10°C and rapid die off below freezing temperatures. Thus, these systems are difficult to maintain in temperate climates. However, it has been suggested to alter hyacinth systems by the introduction of the pennywort (*hydrocotyle umbellata*) during the colder months of the year. Pennyworts may provide an equivalent level of treatment and are more resilient to winter temperatures. No data exist on the performance or cost effectiveness of such systems (Brix, 1993).

Emergent macrophyte based treatment systems utilize plants with an extensive root and rhizome system. Macrophytes typically used are the common reed (*phragmites australis*), the cattail (*typha latifolia*), and the bulrush (*scirpus lacustris*). The three different designs of emergent macrophyte systems include surface flow, horizontal subsurface flow, and vertical subsurface flow. Typical construction of each wetland cell is three to five meters

wide by 100 meters long. The bed material of soil, sand, and gravel is designed according to the desired hydraulic conductivity to accommodate the system flowrate. Oxygen is transferred to the soil zone by the macrophyte root and rhizome. Experience obtained thus far shows a level of treatment for BOD and SS compatible to conventional technologies. Nitrification is difficult to predict or model, as these systems may not provide sufficient oxygen. However, the limited data for vertical flow systems show acceptable performance for BOD, ammonia, and phosphorus. Maintaining continuous flow is difficult, as plant growth and clogging of the porous bed media increases over time. Vertical flow systems operating in a batch scheme of wetting and drying have shown improved efficiency (Brix, 1993).

Submerged macrophyte systems involve the use of plants with photosynthetic tissues completely submerged in the wastewater. These systems only operate well for waters with low concentrations of BOD and high DO concentrations. This type of system is typically used as a polishing step in a multi-stage system. The submerged macrophytic plants reduce the availability of dissolved inorganic carbon and increase the DO. This causes an increase in pH, creating optimal conditions for ammonia volatilization, chemical precipitation of phosphorus, and mineralization and settling of organic matter. The most promising macrophytes thus far include the elodea (*elodea nuttallii* and *canadensis*), hornwort (*ceratophyllum demersum*), hydrilla (*hydrilla verticillata*), and egeria (*egeria densa*) (Brix, 1993).

Approximately half the existing wetland treatment systems in North America have been characterized and compiled into a database by Knight et al, 1993. Preliminary findings of the study found the following average removal efficiencies for constructed wetland treatment systems: BOD₅, 73 percent; TSS, 69 percent; NH₃-N, 44 percent; TN, 64 percent; and TP, 55 percent. The average flowrate of the examined systems exceeded 100,000 gallons per day. In addition, regression analyses compiled from the database sought to model total nitrogen removal as a function of the hydraulic loading rate. The results show that nitrogen removal efficiency is highly sensitive to hydraulic loading rate with a significant decline in efficiency for loadings exceeding 20 kg/ha/d (Knight et al., 1993). Theoretical maximum

removal rates for nitrogen in wetland treatment systems have been computed to be 45 kg/ha/d (Tchobanoglous and Schroeder, 1987). A small scale pilot study treated pulp mill wastewater with an emergent macrophyte system and found maximum ammonia removal of 70 percent with a 24-hour detention time, 62 percent at 15 hours, and 21 percent at six hours (Thut, 1993).

Application of wetland treatment systems to landfill leachate treatment is limited. A three year USGS study at the Thomkins County landfill near Ithaca, NY, treated 2000 L/d of leachate in a small scale emergent macrophyte subsurface flow system. Maximum ammonium removal was 70.8 percent, but was greatly reduced during cold temperatures to an extent that a five percent increase occurred and was attributed to desorption of NH_4^+ from the sediments. Hydraulic residence times were 15 days in the subsurface flow cells. The study concluded that cation exchange and microbial activity in the sediment removes ammonium, but removal is appreciably decreased during the winter months (Surface et al., 1993). A landfill in Escambia County, Florida has utilized a series of ten constructed surface flow emergent growth systems for the treatment of leachate in a closed loop design. The series of leachate treatment operators include a primary treatment basin or lagoon, compost application, wetlands treatment, and the reapplication of treated leachate to the landfill. The system is currently being monitored for effectiveness and its results are inconclusive (Martin et al., 1993). Another pilot test in British Columbia sought to treat landfill leachate for discharge to surface water using a marsh system of surface and subsurface flow. The system bedding material was seeded with waste activated sludge from a POTW to enhance nitrification-denitrification. The maximum observed reduction of ammonia nitrogen was 75 percent. These results are considered inconclusive, as the study was not conducted during winter months (Hunter et al., 1993).

One successful, full-scale application of a wetland treatment technology occurred at a landfill in Brookings, South Dakota. In 1967, a trench was installed downgradient of the landfill to intercept a leachate plume. The trench was excavated to ten meters in depth to a thick clay till deposit serving as a hydraulic boundary. Depth to groundwater ranged from three to four meters. Groundwater degraded by the presence of the leachate plume was

allowed to infiltrate into the excavated trench. While in the trench, leachate constituents were reduced by various biological, chemical, and physical processes including dilution, gas exchange/interchange, metal precipitation, and aerobic and anaerobic biological activity. Subsequent growth of algal and photosynthetic plants in the trench also enhanced treatment by biomass uptake and raising the ambient pH, influencing biodegradation and metals precipitation. The water was then allowed to infiltrate back into the surficial aquifer. Overall water quality of groundwater leaving the trench was found to be comparable to that of downgradient control points. In 1987, an EPA Superfund field investigation team recommended no further action at the site (Hammer, 1989).

Application of a wetland treatment system to the WRL site is limited. Findings from literature and results from pilot-scale facilities are inconsistent and do not offer a reliable means of predicting the success of a wetland treatment system. It is highly improbable that a wetland treatment system alone would provide a sufficient level of treatment to meet NPDES discharge criteria for ammonia nitrogen. In addition, the land requirements for any wetland treatment system are expected to be large. Decreased efficiency is also expected during the winter months. Based on theoretical nitrogen removal rates from Tchobanoglous, 1987, an area of 11 to 14 acres would be required at the WRL site. A more conservative estimate would be 1.5 to two times this amount. A wetland treatment system may be useful if applied in conjunction with other treatment processes such as a biological trickling filter or equivalent system. The wetland could serve as a polishing step after treatment in other processes has occurred. For example, at a flowrate of 450 gpm and a 24-hour detention time, the required size of a lagoon impoundment would be 86619 cubic feet (e.g. 10-foot depth, 105-foot diameter). Larger flowrates or longer detention times (up to 15 days) would increase the size and cost of an impoundment.

Another application of a wetland treatment system potentially applicable to the WRL site is an intercepting trench similar to the one used at the Brookings, SD landfill. Such a system would not be designed to meet NPDES standards nor to discharge water to Kilbuck Creek. The trench could be excavated to a depth to intercept groundwater and/or to accept limited amounts of extracted and pretreated groundwater between the landfill and Kilbuck

Creek. Groundwater in the trench would undergo biological treatment from microbes and introduced free floating and submerged macrophytes. Induced aeration of the upper depths of the water column would most likely aid the progress of aerobic biological activity. The trench could be designed to provide the appropriate hydraulic residence time and allow treated water to infiltrate back into the surficial aquifer of unconsolidated sediments. The infiltration of treated water will provide dilution of surrounding conditions. The expected result would be the improvement of downgradient groundwater quality. Major design considerations include flowrate, detention time, and hydraulic conductivity of trench materials and surrounding native soils. Modeling of these parameters may facilitate the derivation of a feasible design. Other considerations include acceptance by regulating agencies and compliance with local and state flood plain construction regulations.

3.1.5 IN-SITU TREATMENT – AIR SPARGING

An air sparging system introduces air beneath the water table for site remediation. The technology of air sparging involves two mechanisms working alone or together: volatilization and biodegradation. Air sparging can be divided into two distinct technologies, in-well aeration and air injection into the aquifer (Hinchee, 1994). In-well aeration is the process of the injection of gas, usually air, into a well, resulting in an in-well airlift pump effect. Air injection which is applicable to this site involves the introduction of air under pressure directly into saturated groundwater to provide oxygen for bioremediation and/or to strip or volatilize the contaminants present in the aquifer.

The most common air sparging process is to inject air into the saturated zone beneath the water table for the purpose of combined volatilization and aerobic biodegradation of contaminants. It is typically used in conjunction with soil vapor extraction (SVE) in the unsaturated zone to eliminate the offsite migration of vapors.

Air injection is an innovative method for remediating organic compounds present in the saturated zone. In the application of this technology, sparging wells are used to inject a hydrocarbon-free gaseous medium into the saturated zone below or within the areas of contamination. Volatile organic compounds dissolved in the groundwater and sorbed onto

aquifer media partition into the advective air phase, effectively simulating an *in-situ* air stripping system. The stripped contaminants are transported in the air phase to the vadose zone, generally within the radius of influence of a standard vapor extraction well. Air injected into aquifer materials migrates as a separate phase, typically in channels. Contaminated soil within those channels would be aerated; however, aquifer material not within these channels would be much less affected. Marley et al. used an SVE/air-sparging system from 1985 to 1989 to successfully treat BTEX contaminated groundwater at approximately 20,000 to 30,000 ppb (benzene at 225 ppb) down to 600 ppb with non-detectable levels of benzene. Pilot- and full-scale system data indicating successful air sparging operation for BTEX and other VOC removal is available in numerous other references.

The major contaminants at the WRL site are dissolved chlorinated volatile organics at low concentrations and leachate derived inorganics. Chlorinated organics present in the groundwater include tetrachloroethene (PCE), trichloroethene (TCE), dichloroethene, and vinyl chloride. They are rated as low biodegradable organics by Leahy et al. (1994), but with high water solubility and Henry's constants. The Henry's constant for benzene is 0.0055 atm-m³/mol, and for PCE, TCE and vinyl chloride are 0.0153, 0.0091 and 0.081 atm-m³/mol, respectively (Pankow et al., 1993), making them suitable for treatment by air sparging systems. The major removal process these contaminants will be volatilization. Using SVE in conjunction with air sparging at the site may not be necessary due to the low levels of VOCs in groundwater. The major inorganic constituent at the site is ammonia, which has a Henry's constant of 2.91×10^{-4} atm-m³/mol (Montgomery, 1991). Volatilization will assist in a moderate reduction of ammonia. Additional reduction of ammonia and other compounds in groundwater at the site will be achieved by the addition of dissolved oxygen and the resulting natural attenuation by biodegradation.

Natural attenuation by intrinsic biodegradation may be considered as an adjunct technology to air sparging. Intrinsic biodegradation is most feasible when contaminant concentrations are low and when the source areas have been eliminated (Johnson et al., 1995). Biodegradation will reduce contaminant concentrations in impacted zones outside of

the radius of influence of the sparging wells. The addition of dissolved oxygen by the sparging system will significantly increase biodegradation rates after air sparging is initiated. The increased activity of indigenous, aerobic microbes which will oxidize ammonia (nitrification) and any remaining organics, is expected to occur in downgradient regions from the sparging system.

Dissolved oxygen also will create favorable redox conditions for the sorption of dissolved metals such as iron (Bjerg et al., 1995). Downgradient regions will be monitored to measure the effectiveness of biodegradation over time. Some of the important monitoring parameters for *in-situ* bioremediation include DO, temperature, pH, alkalinity, total organic carbon (TOC), total Kjeldahl nitrogen (TKN), and ammonia (Johnson et al., 1995). In addition, downgradient monitoring wells will be sampled for the constituents of interest to affirm that a reduction of contaminant mass is occurring.

3.2 DETAILED ANALYSIS OF GROUNDWATER REMEDIATION ALTERNATIVES

This section analyzes the ability of the corrective measures to meet appropriate regulatory requirements and expectations. The assessment of the corrective measures focuses specifically on the minimum requirements outlined in 35 IAC 811.324(d)(1)-(4). Examples of these requirements include performance, reliability, ease of implementation, completion time, and cost. Table 3.2-1 is used to screen the alternatives with respect to these criteria in addition to other relevant factors. Table 3.2-2 presents the estimated capital and annual operation and maintenance costs associated with each corrective measure. A detailed cost analysis of air stripping was previously provided (GeoTrans, 1995c). Costs associated with the air sparging alternative are illustrated in Section 4.5.

Table 3.2-1 shows that all of the corrective measures will perform removal of the contaminants of concern. However, only the measures associated with a pump-and-treat system will provide hydraulic containment of the landfill area. All of the technologies except wetlands treatment are more likely than not to meet the required efficiency and performance

Table 3.2-1. Detailed analysis of corrective action measures for WRL.

Analysis Factor	<i>Ex-situ</i> Treatment Air Stripping	<i>Ex-situ</i> Treatment Trickling Filter/ Fluidized Bed Reactor	<i>Ex-situ</i> Treatment Sequencing Batch Reactor	<i>Ex-situ</i> Treatment Constructed Wetlands Treatment	<i>In-situ</i> Treatment Air Sparging
Criteria: Performance (811.324d(1))					
Magnitude of The Remaining Risks	Low, contaminants of concern are removed and migration is controlled	Low, contaminants of concern are removed and migration is controlled	Low, contaminants of concern are removed and migration is controlled	Low, contaminants of concern are removed and migration is controlled	Moderately low, contaminants of concern are removed
Remaining sources of risks:					
• Treatment residuals	No risk	No risk	No risk	No risk	No risk
• Untreated residual contamination	Low risk	Low risk	Low risk	Low risk	Low risk
• 5-year review required?	Yes, to monitor spread of contamination and assess treatment system performance	Yes, to monitor spread of contamination and assess treatment system performance	Yes, to monitor spread of contamination and assess treatment system performance	Yes, to monitor spread of contamination and assess treatment system performance	Yes, to monitor spread of contamination and assess treatment system performance. Potential short term (i.e., every year) reviews are also needed.
Criteria: Reliability (811.324d(1))					
• Likelihood that technology will meet required process efficiencies of performance specifications	Very likely	Very likely	Very likely	Moderately likely	Very likely
• Type and degree of long-term management	Extraction and treatment system O&M. Analysis of groundwater monitoring data. Moderate maintenance.	Extraction and treatment system O&M. Analysis of groundwater monitoring data. Moderate maintenance.	Extraction and treatment system O&M. Analysis of groundwater monitoring data. Moderate maintenance.	Extraction and treatment system O&M. Analysis of groundwater monitoring data. Moderate to high maintenance.	Sampling and analysis of groundwater. Collection and review of monitoring data. Moderate maintenance.
• Long-term monitoring required?	Yes. Air, process, and groundwater monitoring.	Yes. Process and groundwater monitoring	Yes. Process and groundwater monitoring	Yes. Process and groundwater monitoring	Yes. Air and groundwater monitoring

Table 3.2-1. Detailed analysis of corrective measures alternatives for WRL (continued).

Analysis Factor	<i>Ex-situ</i> Treatment Air Stripping	<i>Ex-situ</i> Treatment Trickling Filter/ Fluidized Bed Reactor	<i>Ex-situ</i> Treatment Sequencing Batch Reactor	<i>Ex-situ</i> Treatment Constructed Wetlands Treatment	<i>In-situ</i> Treatment Air Sparging
• Operation and maintenance functions to be performed	Equipment maintenance, analysis of influent/effluent water; O&M related to long-term monitoring. Emissions monitoring.	Equipment maintenance, analysis of influent/effluent water; O&M related to long-term monitoring.	Equipment maintenance, analysis of influent/effluent water; O&M related to long-term monitoring.	Equipment maintenance, analysis of influent/effluent water; O&M related to long-term monitoring.	Equipment maintenance, O&M related to long-term monitoring. Emissions monitoring.
• Difficulties and uncertainties associated with long-term operation and maintenance	System efficiency for long-term performance is uncertain. Scaling.	System efficiency for long-term performance is uncertain. Bed clogging.	System efficiency for long-term performance is uncertain.	System efficiency for long-term performance is uncertain.	System efficiency for long-term performance is uncertain.
• Potential need for replacement of technical components	Low	Low	Low	Moderately low	Low
• Degree of confidence that control can adequately handle potential problems	High	High	High	Moderate	High
• Uncertainties associated with land disposal of residual and untreated wastes	None	None	None	None	None
Criteria: Reduction of Toxicity, Mobility, or Volume Through Treatment					
Treatment process and remedy:					
• Principal threats addressed?	Yes	Yes	Yes	Yes	Yes
• Special requirements for the treatment process?	No	No	No	Yes. Large land area.	No
Type and quantity of treatment residuals:					
• Quantities and characteristics of residuals	Filter cake, if filter press is used. Estimated 8.5 cubic yards per day.	Filter cake, if filter press is used. Estimated 8.5 cubic yards per day.	Filter cake, if filter press is used. Estimated 8.5 cubic yards per day.	Harvested (replaced) macrophytic plants.	Spent carbon, if GAC is used for emission control.
• Risks posed by treatment residuals	Low	Low	Low	Very low	Very low

Table 3.2-1. Detailed analysis of corrective measures alternatives for WRL (continued).

Analysis Factor	<i>Ex-situ</i> Treatment Air Stripping	<i>Ex-situ</i> Treatment Trickling Filter/ Fluidized Bed Reactor	<i>Ex-situ</i> Treatment Sequencing Batch Reactor	<i>Ex-situ</i> Treatment Constructed Wetlands Treatment	<i>In-situ</i> Treatment Air Sparging
Statutory preference for treatment as a principal element:					
• Are principal threats addressed within the scope of action?	Yes	Yes	Yes	Yes	Yes
• Is treatment used to reduce inherent hazards posed by principal threats at the site?	Yes	Yes	Yes	Yes	Yes
Criteria: Implementation (811.324d(1))					
• Relative ease of implementation	High	High	High	Moderate	High
Criteria: Safety impacts/short-term effectiveness (811.324d(1))					
Protection of community during remedial actions:					
• Risks to the community that must be addressed during remedial action	Potential dust generation during well and treatment system installation.	Potential dust generation during well and treatment system installation.	Potential dust generation during well and treatment system installation.	Potential dust generation during site preparation and well installation.	Potential dust generation during well and treatment system installation.
• How will the risks to the community be addressed and mitigated?	Engineering control (dust suppression)	Engineering control (dust suppression)	Engineering control (dust suppression)	Engineering control (dust suppression)	Engineering control (dust suppression)
• What risks remain to the community that cannot be readily controlled?	None	None	None	None	None
Protection of workers during remedial actions:					
• Risks to the workers that must be addressed during remedial action	Health and safety during construction	Health and safety during construction	Health and safety during construction	Health and safety during construction	Health and safety during construction

Table 3.2-1. Detailed analysis of corrective measures alternatives for WRL (continued).

Analysis Factor	<i>Ex-situ</i> Treatment Air Stripping	<i>Ex-situ</i> Treatment Trickling Filter/ Fluidized Bed Reactor	<i>Ex-situ</i> Treatment Sequencing Batch Reactor	<i>Ex-situ</i> Treatment Constructed Wetlands Treatment	<i>In-situ</i> Treatment Air Sparging
• How will the risks to the workers be addressed and mitigated?	Proper engineering controls and personal protective equipment	Proper engineering controls and personal protective equipment	Proper engineering controls and personal protective equipment	Proper engineering controls and personal protective equipment	Proper engineering controls and personal protective equipment
• What risks remain to the workers that cannot be readily controlled?	None	None	None	None	None
Criteria: Cross media impact (811.324d(1))					
• Cross media impacts?	Yes. Results in discharge to surface water	Yes. Results in discharge to surface water	Yes. Results in discharge to surface water	Yes. Results in discharge to surface water	Yes, between groundwater and soil vapor
• How will cross media impacts be addressed?	NPDES Permit	NPDES Permit	NPDES Permit	NPDES Permit	Soil vapor extraction as part of air sparging system.
Criteria: Control of exposure to any residual contamination (811.324d(1))					
• Potential exposure to residual contamination on site	None	None	None	None	None
Criteria: Technical feasibility					
Ability to construct and operate technology:					
• Difficulties associated with construction	None	None	None	Large area of land (earthwork) needed to construct wetland.	None
• Uncertainties related to construction	None	None	None	Landfill is located on the flood plain	None
Reliability of technology:					
• Likelihood that technical problems will lead to delays	Not likely	Not likely	Not likely	Moderately likely	Not likely

Table 3.2-1. Detailed analysis of corrective measures alternatives for WRL (continued).

Analysis Factor	<i>Ex-situ</i> Treatment Air Stripping	<i>Ex-situ</i> Treatment Trickling Filter/ Fluidized Bed Reactor	<i>Ex-situ</i> Treatment Sequencing Batch Reactor	<i>Ex-situ</i> Treatment Constructed Wetlands Treatment	<i>In-situ</i> Treatment Air Sparging
Ease of undertaking additional corrective measures:					
• Likely future corrective measures	None	None	None	None	None
• Difficulties in implementation of additional corrective measures	None	None	None	None	None
Monitoring considerations:					
• Do migration or exposure pathways exist that cannot be monitored adequately?	No	No	No	No	No
• What risks or exposures exist should monitoring be insufficient to detect failure?	None, monitoring will be sufficient to detect failure.	None, monitoring will be sufficient to detect failure.	None, monitoring will be sufficient to detect failure.	None, monitoring will be sufficient to detect failure.	None, monitoring will be sufficient to detect failure.
Criteria: Availability of services and materials					
Availability of treatment, storage capacity, and disposal services:					
• Availability of treatment, storage capacity, and disposal services	Available	Available	Available	Available	Available
• Additional capacity necessary?	No	No	No	Possibly	No
• Additional provisions required to ensure the needed additional capacity	None	None	None	None	None
Availability of necessary equipment and specialists:					
• Are the necessary equipment and specialists available?	Yes	Yes	Yes	Yes	Yes

Table 3.2-1. Detailed analysis of corrective measures alternatives for WRL (continued).

Analysis Factor	Ex-situ Treatment Air Stripping	Ex-situ Treatment Trickling Filter/ Fluidized Bed Reactor	Ex-situ Treatment Sequencing Batch Reactor	Ex-situ Treatment Constructed Wetlands Treatment	In-situ Treatment Air Sparging
• What additional equipment and specialists are required?	None	None	None	None	None
• Does the lack of equipment and specialists prevent implementation?	N/A	N/A	N/A	N/A	N/A
• What additional provisions are required to ensure the needed equipment and specialists?	N/A	N/A	N/A	N/A	N/A
Availability of prospective technologies:					
• Will technologies require further development before they can be applied full-scale to the type of waste at the site?	No	No	No	No	No
• Will more than one vendor be available to provide a competitive bid?	Yes	Yes	Yes	Yes	Yes
Criteria: Time requirement (811.324d(2))					
Time until response actions are achieved:					
• Time until corrective measure objectives are achieved	10 years (estimated)	10 years (estimated)	10 years (estimated)	10 years (estimated)	10 years (estimated)
Criteria: Cost (811.324d(3))					
• Capital Cost	High	High	High	Undetermined	Moderate
• Annual O&M Cost	Very high	Moderate to High	High	Undetermined	Low
Criteria: Institutional requirement (811.324d(4))					
Potential permit requirements:					
• NPDES	Yes	Yes	Yes	Yes	No

Table 3.2-1. Detailed analysis of corrective measures alternatives for WRL (continued).

Analysis Factor	<i>Ex-situ</i> Treatment Air Stripping	<i>Ex-situ</i> Treatment Trickling Filter/ Fluidized Bed Reactor	<i>Ex-situ</i> Treatment Sequencing Batch Reactor	<i>Ex-situ</i> Treatment Constructed Wetlands Treatment	<i>In-situ</i> Treatment Air Sparging
• Air notification	Yes	No	No	No	Yes
• Building	Yes	Yes	Yes	Yes	Yes
• Electric	Yes	Yes	Yes	Yes	Yes
• Sediment and erosion control plan	Yes	Yes	Yes	Yes	Yes

Note: NA = Not applicable

Table 3.2-2. Summary of costs of corrective action measures for WRL Site.

	<i>Ex-Situ</i> Treatment- Air Sparging	<i>Ex-Situ</i> Treatment - Trickling Filter/Fluidized Bed Reactor	<i>Ex-Situ</i> Treatment - Sequencing Batch Reactor	<i>In-situ</i> Treatment - Air Sparging
Total Capital Cost for Groundwater Extraction System	\$111,500	\$111,500	\$111,500	N/A
Total Capital Cost	\$2,374,800	\$530,000/ \$1,700,000	\$670,000	\$314,300
Capital Subtotal	\$2,486,300	\$641,500/ \$1,811,500	\$781,500	\$314,300
Contingencies Bid and Scope (15%) Construction Service, Startup	\$622,945	\$350,000/ \$522,000	\$367,225	\$107,145
Total Construction Cost	\$3,109,245	\$991,500/ \$2,333,500	\$1,148,725	\$421,445
Additional Contingencies	\$350,000 (Solids removal and magnetic sequestering)	\$300,000 (Solids removal)/ \$300,000 (Solids removal)	\$300,000 (Solids removal after sequencing batch reactor)	Off-gas treatment and enhanced oxygen supply
Total Annual Operation and Maintenance	\$786,428	\$300,000/ \$85,000	\$360,000	\$36,800
Annual O&M Estimate of Contingencies	\$248,280 (Filter cake disposal, electrical power and maintaining)	\$248,280 (Filter cake disposal, electrical power, oxygen supply and maintaining)	\$248,280 (Filter cake disposal, electrical power and maintaining)	Minimal GAC replacement and enhanced oxygen supply

N/A - Not available

expectations and have a high relative ease of implementation. None of the proposed alternatives have any adverse health and safety impacts to workers or the community or allow any exposure to residual site contamination. Air sparging contains the most acceptable cross media impact between groundwater and soil vapor. This impact is inherently part of the process of air sparging and will be mitigated by concurrent soil vapor extraction. Corrective measures associated with pump-and-treat are designed to discharge to a nearby surface water body, Kilbuck Creek, under an NPDES permit. All alternatives require significant permit requirements. In particular, the *ex-situ* corrective measures will require a NPDES permit for the discharge of treated groundwater to Kilbuck Creek. The completion time for all alternatives is estimated at ten years. Table 3.2-2 illustrates that the costs associated with air sparging are at least half to an order of magnitude lower than those associated with the other corrective measures. Thus, air sparging is the most cost effective alternative.

In-situ air injection (air sparging) and associated enhanced biodegradation are the remedial alternatives recommended for the WRL site. The selection is based on hydrogeologic conditions, the chemical characteristics of the major contaminants, and the ability of remedy to attain the requirements of 35 IAC 811.324(d). The relatively homogeneous sands and high hydraulic conductivity have proven ideal for effective operation of air sparging systems at similar sites. The chlorinated volatile contaminants at the downgradient perimeter of the landfill are in relatively low concentrations and should be effectively removed. Additionally, the increasing DO levels in this oxygen-deficient area will improve general water quality by promoting aerobic biodegradation. While the air sparging system is operating, elimination of the source area will be achieved by leachate collection. The installation of a composite geomembrane cover is described in the application for the significant modification to the existing unit (SIG MOD). The source area elimination, enhanced bioremediation, and change in redox potential will lower concentrations of elevated constituents over time in previously impacted zones downgradient of the landfill. Section 4 will demonstrate that air sparging meets the requirements of 35 IAC 811.325 as a selected remedy.

If air sparging with enhanced biodegradation is not an acceptable remedial alternative since it does not include extraction of groundwater as specified in the ROD, the most cost-effective, reliable treatment method remaining would be the *ex-situ* FBR-GAC treatment technology to remove ammonia and VOCs from extracted groundwater. This alternative has significantly higher associated costs than the *in-situ* air sparging alternative. Additionally, a pump-and-treat system would not directly enhance aquifer conditions with DO addition as will the air sparging system. Rather, remediation would be accomplished by groundwater unaffected by the landfill constituents flowing into the area downgradient of the landfill enhancing biodegradation/attenuation.

3.3 SELECTED GROUNDWATER REMEDIATION SYSTEM

3.3.1 REMEDY REQUIREMENTS

Section 3 presented a corrective action measure assessment in accordance with IAC Section 811.324. This section will demonstrate that the selected remedy of *in-situ* air sparging in combination with the landfill cap and upgraded leachate extraction system already planned for the site meets the requirements set forth in IAC Section 811.325. Specifically, the selected remedy must meet the following requirements: (1) protection of human health and the environment; (2) attainment of groundwater quality standards (811.320); (3) control the sources of release; and (4) compliance with standards for management of wastes (811.326(d)). The degree to which the selected remedy addresses these requirements is discussed below.

Protection of Human Health and the Environment (811.325(b)(1))

Risks to human health and the environment are reduced by the final landfill cap and upgraded leachate extraction system. These actions will minimize leachate production and subsequent migration into groundwater through reduced infiltration and enhanced extraction. Pre-existing contaminated and migrating groundwater will be directly addressed by air

sparging. Air sparging will reduce contaminant concentrations to acceptable levels, thus minimizing risks to downgradient potential human and environmental receptors.

Attainment of Groundwater Quality Standards (811.325(b)(2))

Groundwater quality standards determined in the Groundwater Monitoring Plan (GeoTrans, 1995a) will be achieved over time by the use of air sparging. Air sparging will directly address VOC contamination and indirectly address other water quality parameters such as ammonia by providing enhanced DO conditions conducive to aerobic biodegradation.

Control of the Source of Release (811.325(b)(3))

The source of release will be controlled by both the final landfill cap and upgraded leachate extraction system. These remedies will concurrently reduce the volume of contaminants released to groundwater to acceptable levels.

Compliance with Waste Management Standards (811.325(b)(4))

The WRL site will continue to operate in a manner consistent with the applicable requirements in 35 IAC 811. Spent carbon from the treatment of sparging off-gases will either be sent to an appropriate off-site disposal facility or returned to the vendor for regeneration.

3.4 DETAILED ASSESSMENT OF SELECTED REMEDY

The evaluation factors described in 35 IAC 811.325(c) were considered in the selection of the site remedy and are discussed individually below.

Long and Short-Term Effectiveness (811.325(c)(1))

The final cap and upgraded leachate extraction system will provide immediate short-term benefits and long-term effectiveness of controlling releases from the source area. Long-term effectiveness is also realized by the air sparging system which will reduce VOC levels

in groundwater and enhance biodegradation of organic and inorganic contaminants in the subsurface.

Magnitude of Risk Reduction (811.325(c)(1)(A))

As previously discussed, risks presented to potential human and environmental receptors will be reduced to acceptable levels by meeting the groundwater AGQS. Residual risks associated with the likelihood of further releases will be minimal due to containment by the cap and leachate extraction system (811.325(c)(1)(B)). Short-term risks are only applicable to site workers during the implementation and operation of the remedial measures (811.325(c)(1)(D)). Appropriate worker health and safety measures during site work will address short-term risks. The selected remedy will not result in any potential for exposure of human or environmental receptors to remaining wastes (811.325(c)(1)(F)). Long-term management of risks will be addressed by the monitoring program and operation and maintenance of the remedial measures (811.325(c)(1)(C)). It is estimated that long-term risk management will proceed for 10 years after the remedial measures are implemented, at which time, full protection is achieved (811.325(c)(1)(E)).

Effectiveness (811.325(c)(2))

The containment measures (capping and leachate extraction) will provide a significant level of effectiveness to control and reduce source releases. The air sparging system will be effective in reducing, and in some cases removing, VOCs from currently contaminated groundwater. Air sparging will also be effective in enhancing water quality conditions to promote aerobic biodegradation of ammonia and other groundwater contaminants.

Implementability (811.325(c)(3))

All of the remedial measures planned for use at the WRL site have no significant difficulties associated with their construction or operation and maintenance. The operational reliability of these measures has proven to be credible at other sites with similar conditions and contaminants. In addition, the measures do not stress the limits of site capacity as

minimal land area is required. Permits associated with the implementation of the measures will likely be obtained through the IEPA and possibly USEPA and will require little coordination with other agencies. Furthermore, specialists, equipment vendors, and contractors required for installation are readily available in the commercial market. Lastly, the owner and operators of the site have the practical technical and economic capabilities to implement the remedial measures (811.325(c)(4)). In accordance with 811.324(e), the results of the corrective measures assessment will be presented in a public meeting tentatively scheduled to be held within 60 days of approval of this preliminary design. The degree to which community concerns are addressed by the selected remedy (811.325(c)(5)) will be evaluated at a future date.

Schedule (811.325(d))

A schedule for remediation activities is discussed in Section 5.6 and was developed considering the relevant factors outlined in 35 IAC811.325(d) including the aforementioned evaluation factors.

4 AIR SPARGING SYSTEM DESIGN

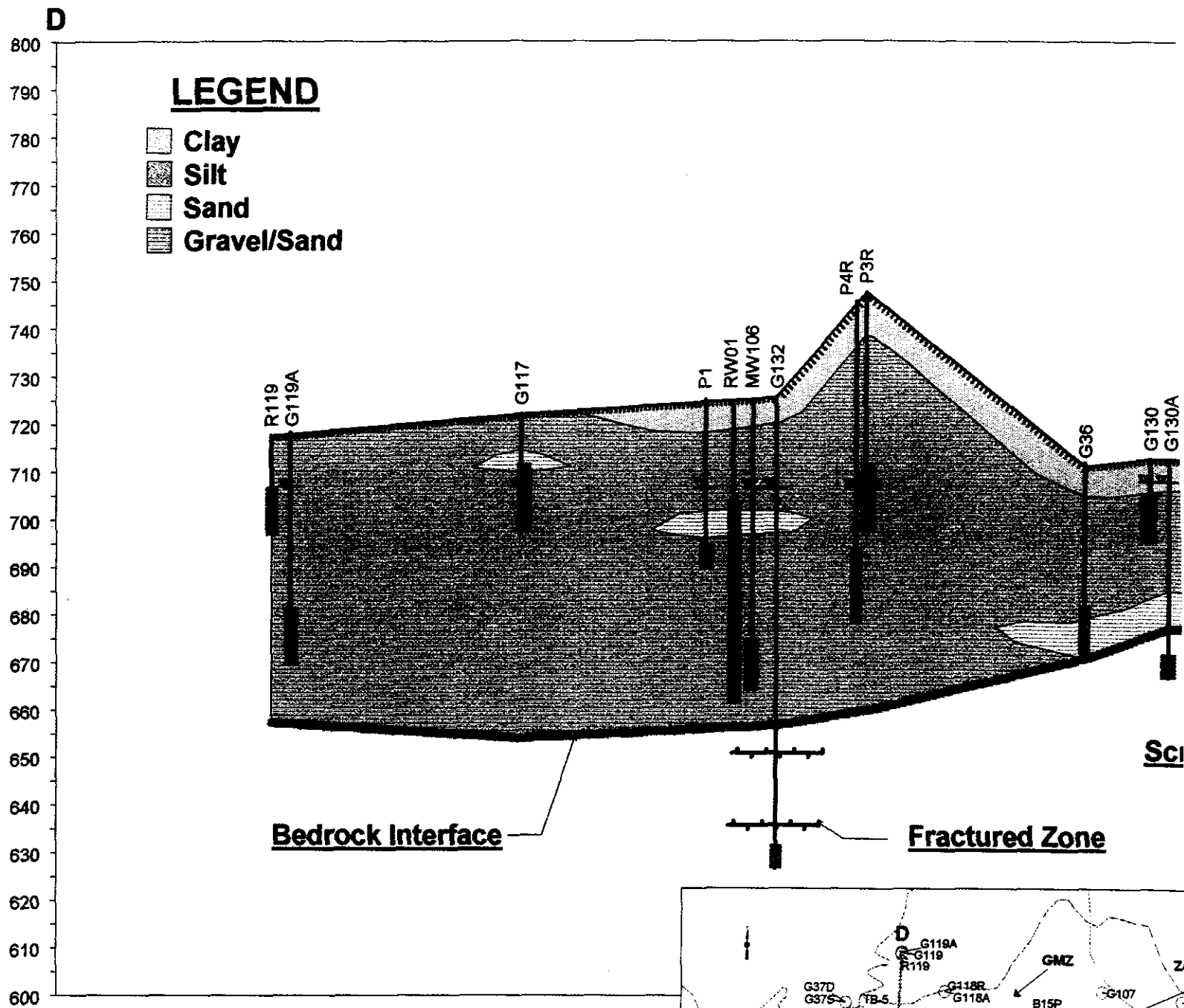
There are numerous criteria that must be considered when designing, installing, and operating an *in-situ* air sparging system. The system has to ensure effective remediation of saturated zone soil and groundwater while preventing the displacement and mobilization of soil-gas vapors or dissolved phase contaminants in the aquifer.

This section will review WRL site information and present the preliminary design of the proposed air sparging and soil vapor extraction system.

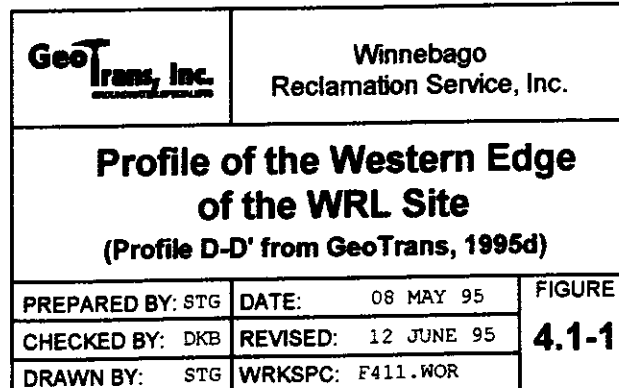
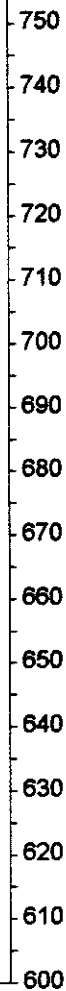
4.1 GEOLOGY/HYDROGEOLOGY OF SITE

Site geology is considered the most important design parameter. Air sparging is generally more effective in coarse-grained, homogeneous soil. Coarse soils have lower air entry pressure requirements and provide a medium for even air distribution, allowing for better mass transfer efficiencies and effective VOC removal. At the west side of the landfill, where sparging wells are proposed, the aquifer is mostly sand and gravel and has a clay layer of about five feet near the surface. The aquifer material and its relative homogeneity are very suitable for air sparging. A geologic cross section map (Figure 4.1-1) illustrates this formation. The hydraulic conductivity of the sand and gravel aquifer is 1500 ft/day with a porosity of 30.4 percent (GeoTrans, 1995d). Based on a recent air sparging field test in a similar hydrogeologic setting as the WRL site (Lundegard, 1995), an estimated radius of influence of up to 15 feet would be established. This favorable geologic setting makes air sparging applicable to the WRL site.

Previous reports presented the geologic and hydrogeologic settings of the WRL site (GeoTrans, 1995a, b and d). A higher permeability, unconsolidated sand and gravel aquifer and lower permeability dolomite bedrock aquifer form the aquifer system beneath the WRL site. Previous reports also show that groundwater in the sand and gravel aquifer generally flows to the west-northwest direction. The aquifer system is recharged in the bedrock uplands with groundwater flowing downward in this area and later flowing back upward into



Note: Groundwater elevations are based on water level data collected on 02/17/95.

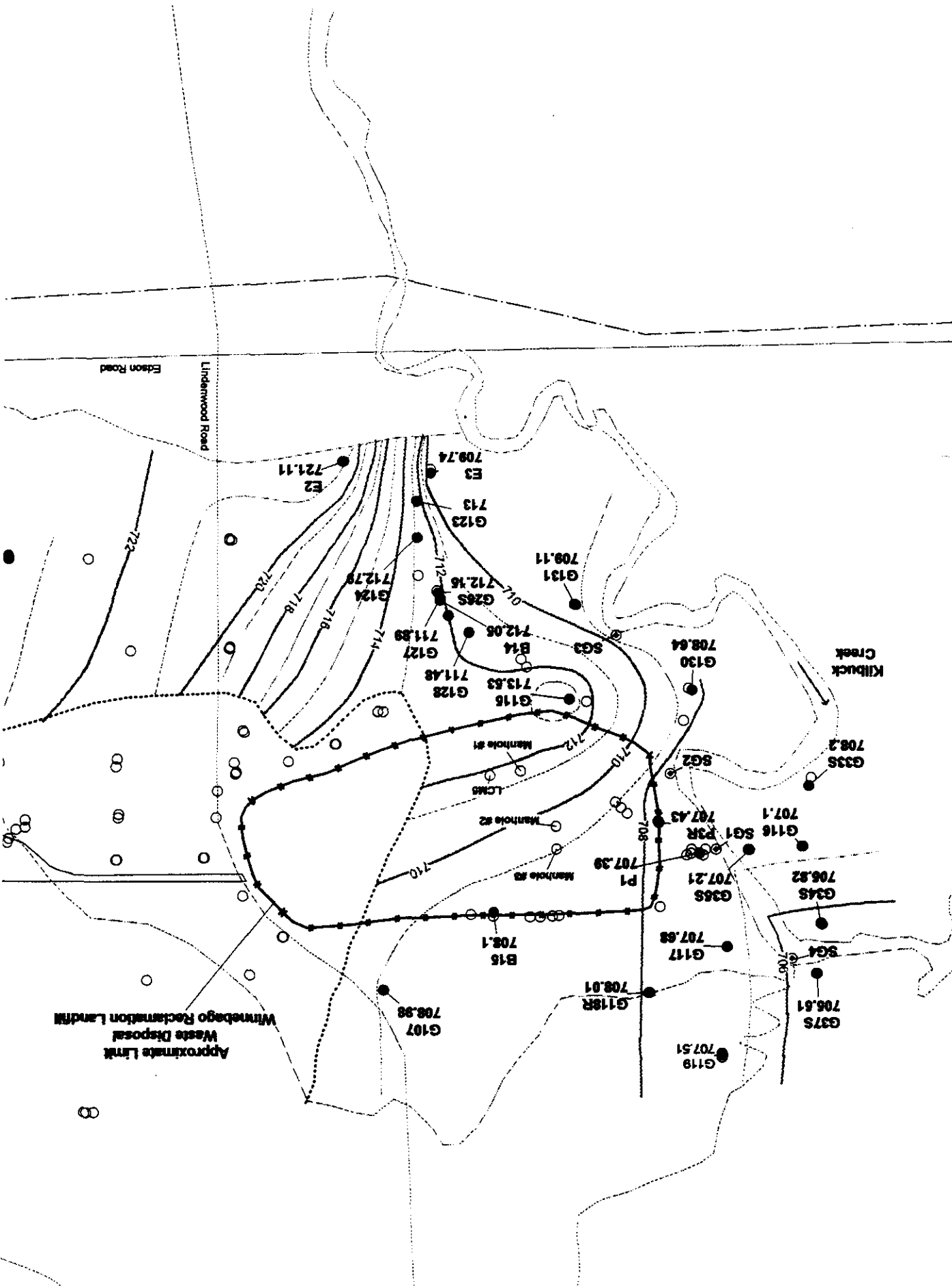


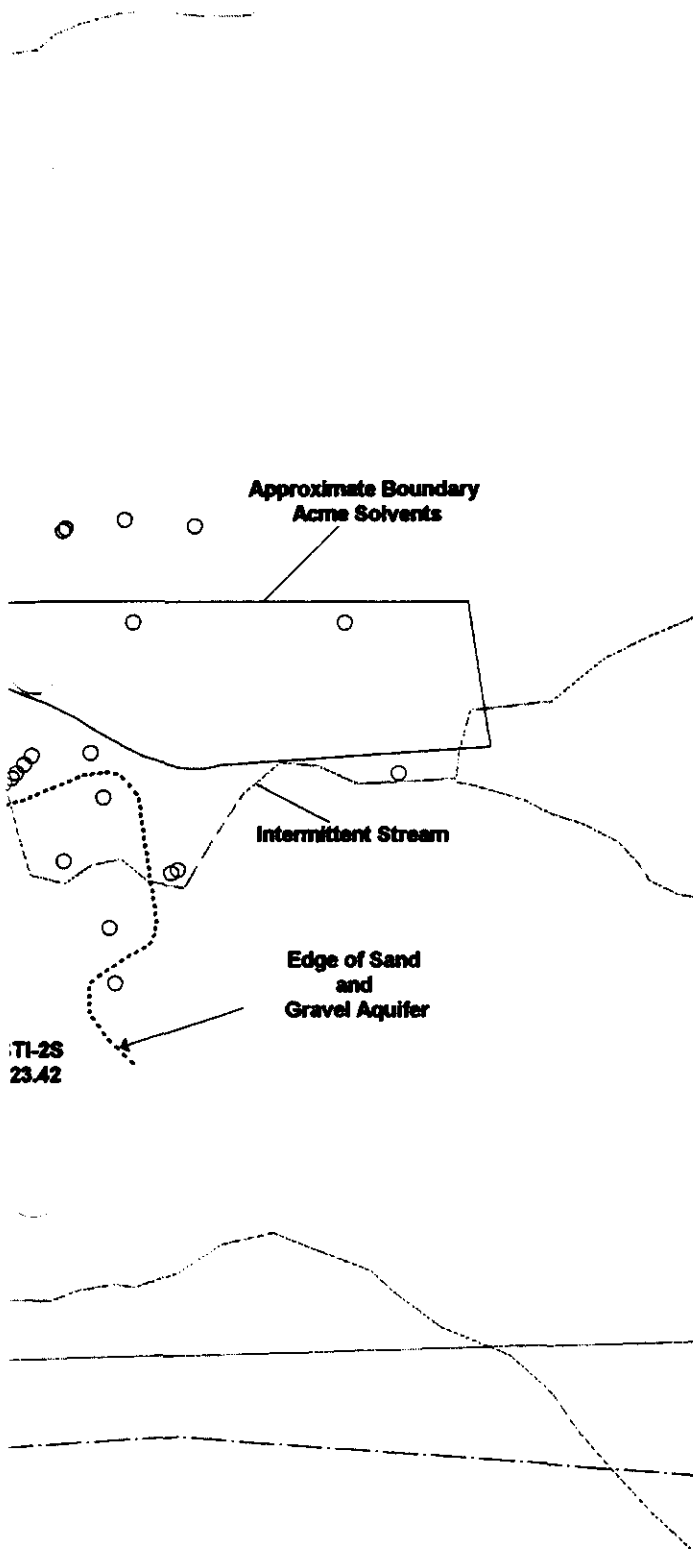
the higher permeability sand and gravel sediments (GeoTrans, 1995a). Shallow groundwater in the unconsolidated sediments discharges to Kilbuck Creek, while deeper groundwater flows beneath the creek and continues toward the west-northwest. Figures 4.1-2 and 4.1-3 illustrate these flow characteristics by groundwater elevations in the upper and lower zone of the sand and gravel aquifer, respectively.

Figures 4.1-4, through 4.1-6, respectively, show the distribution of PCE, TCE, and ammonia. It is apparent that ammonia is transported through the aquifer and either discharges into Kilbuck Creek or flows under the creek toward the northwest. Monitor wells P3R, P4R and G132, which are on west side of the landfill, are the only wells showing historically elevated concentrations of VOCs. Across the creek, wells G35S and G116A have also shown slightly elevated levels of VOCs. These sample results correspond with both observed data and the calibrated flow model, showing that impacted groundwater at the Acme Solvent Superfund site east of the WRL site flows upward from bedrock into the sand and gravel aquifer. It is anticipated that the migration of VOCs from the Acme Solvent site will diminish once the groundwater remediation system at this site is fully operational in the summer of 1995. The sparging wells located on the west side of the landfill would intercept these contaminant plumes and remove both VOCs and ammonia from the WRL groundwater.

In June 1995, field measurements of dissolved oxygen, pH, and specific conductance were measured at 40 monitoring wells across the WRL site and at Kilbuck Creek. Water was removed from these monitoring points prior to sampling. The field measured values of dissolved oxygen, pH, and specific conductance are summarized in Table 4.1-1.

Figures 4.1-7 through 4.1-9 show observed June 1995 dissolved oxygen concentrations for the upper and lower zones of the sand and gravel aquifer, and the bedrock aquifer. It is apparent that there is a region of low dissolved oxygen concentrations in monitoring wells downgradient of the landfill. These wells have had historically high inorganic compound concentrations. The low dissolved oxygen concentrations in these wells, and corresponding elevated concentrations of compounds that are indicators of a lower redox potential (iron (dissolved), sulfate, and ammonia), suggest that oxygen is a limiting factor for aerobic metabolism by natural bacteria. In addition, the presence of elevated





0 680 1360

feet



Winnebago Reclamation Services

LEGEND

- Monitoring Well
- G101 Well I.D. and Groundwater
750.90 Elevation
- SG3 Stream Gage I.D.
- ~~~~~ 750 Groundwater Elevation Contours
- ~~~~~ Intermittent Stream
- ~~~~~ Edge of Sand and Gravel Aquifer

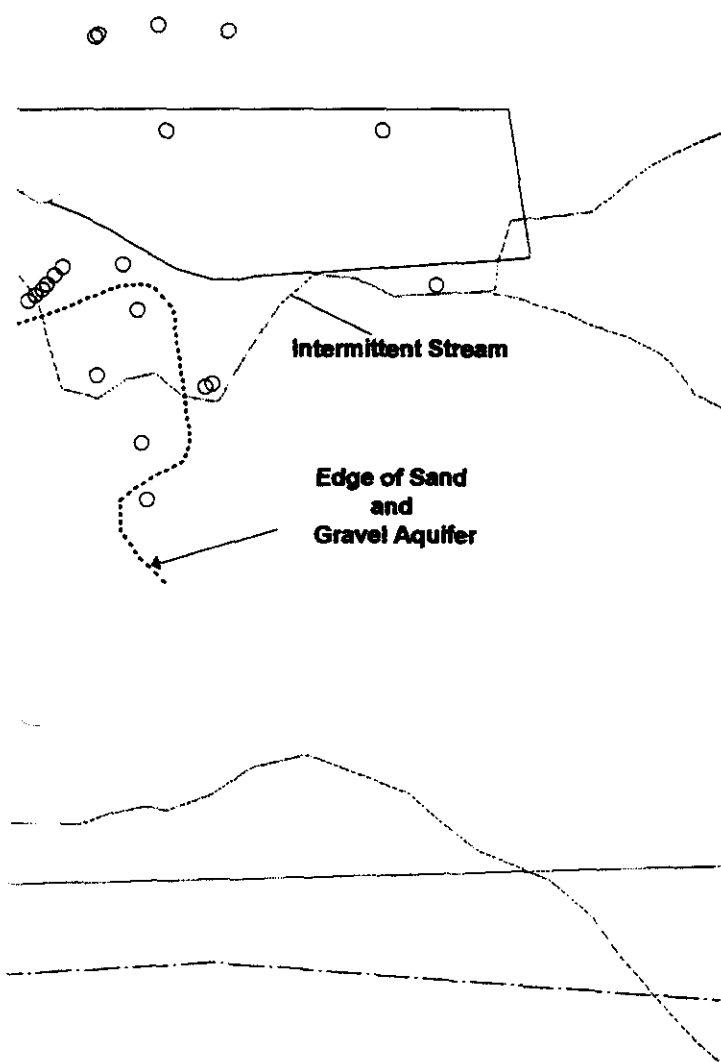
TITLE: Groundwater Elevation Contours
Upper Zone of the Sand and Gravel Aquifer
(4/25/95)

LOCATION: Winnebago Reclamation Services, Rockford, IL.

GeoTrans, Inc.

CHECKED:	DB
DRAFTED:	ELA
FILE:	F412.WOR
	04/25/95

FIGURE:
4.1-2



0 680 1360
feet

N



Winnebago Reclamation Services

LEGEND

- Monitoring Well
- G101 Well I.D. and Groundwater 750.90 Elevation
- SG3 Stream Gage I.D.
- ~~~~~ 750 Groundwater Elevation Contours
- ~~~~~ Intermittent Stream
- ~~~~~ Edge of Sand and Gravel Aquifer

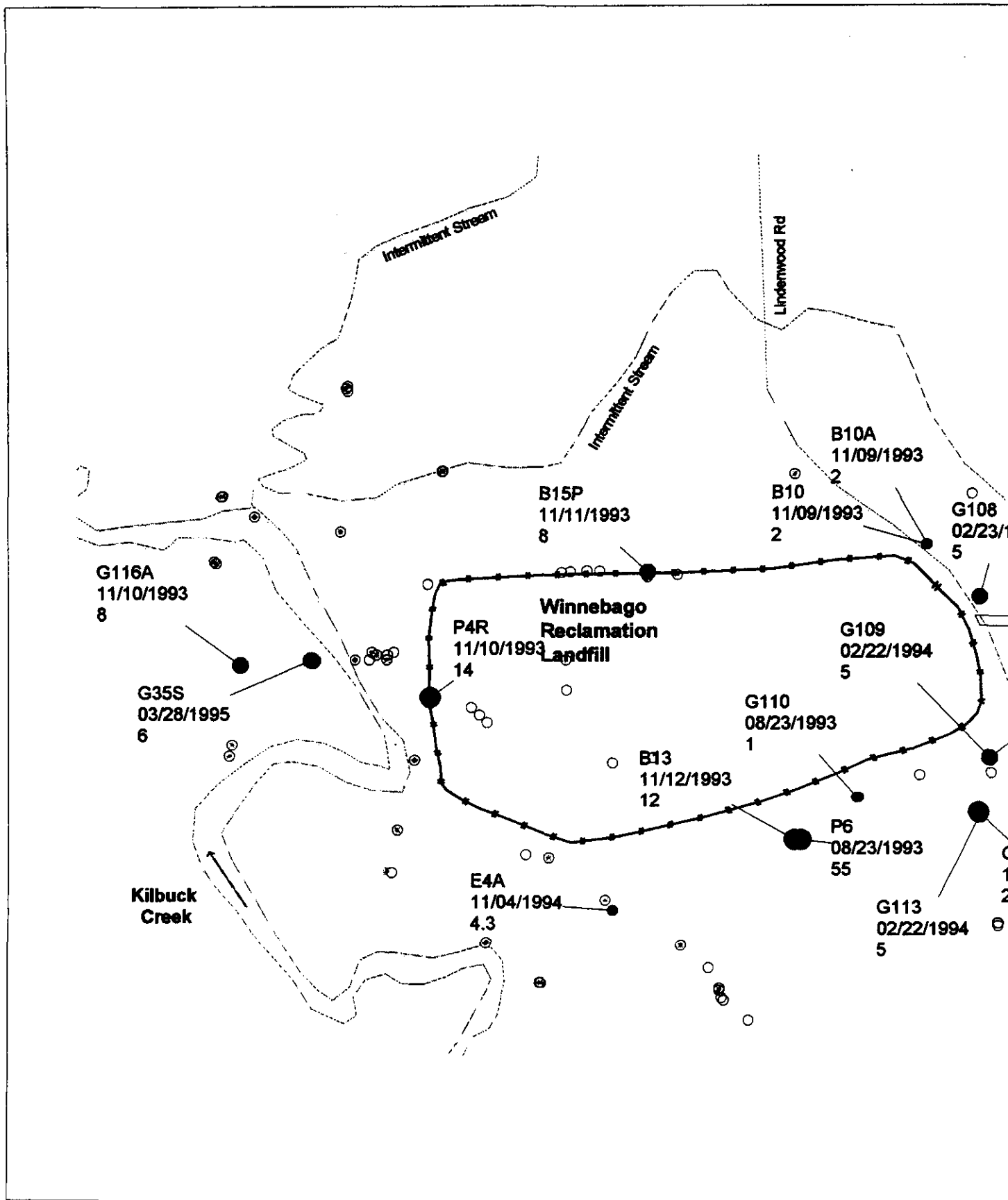
TITLE: Groundwater Elevation Contours
Lower Zone of the Sand and Gravel Aquifer
(4/25/95)

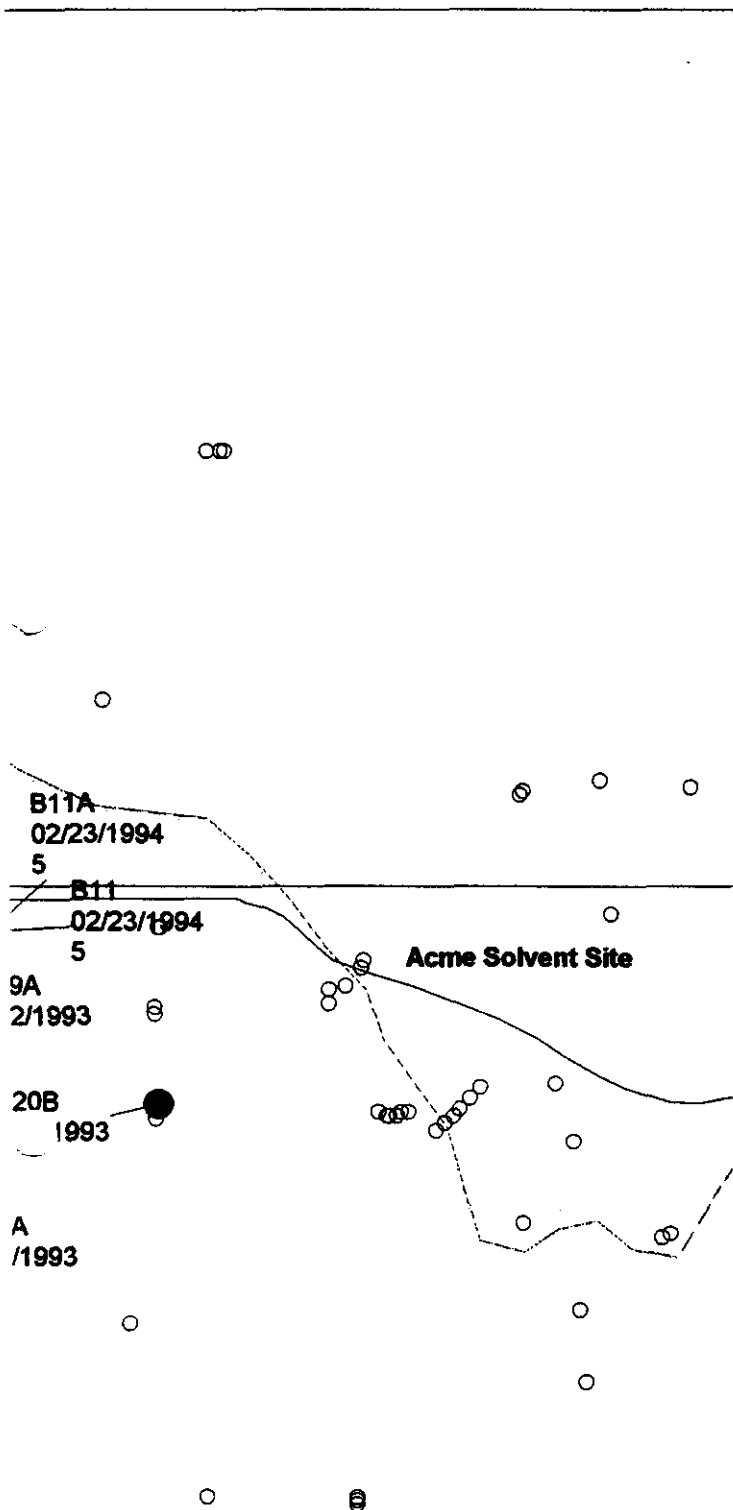
LOCATION: Winnebago Reclamation Services, Rockford, IL.

GeoTrans, Inc.
GROUNDWATER SPECIALISTS

CHECKED: DB
DRAFTED: FLA
FILE: F413.WOR
DATE: 9 MAY 1995

FIGURE:
4.1-3





PCE (1993-95)

Concentrations in ug/L

- > 10 (11)
- 5 to 10 (21)
- 1 to 5 (15)
- < 1 (0)
- nd (108)

Note: Labels indicate date and value of maximum detection.



Concentration of PCE
at WRL Site Wells
(Current GeoTrans Database)

GeoTrans, Inc.
GROUNDWATER SPECIALISTS

Prepared by: MPM

Date: 5/12/95

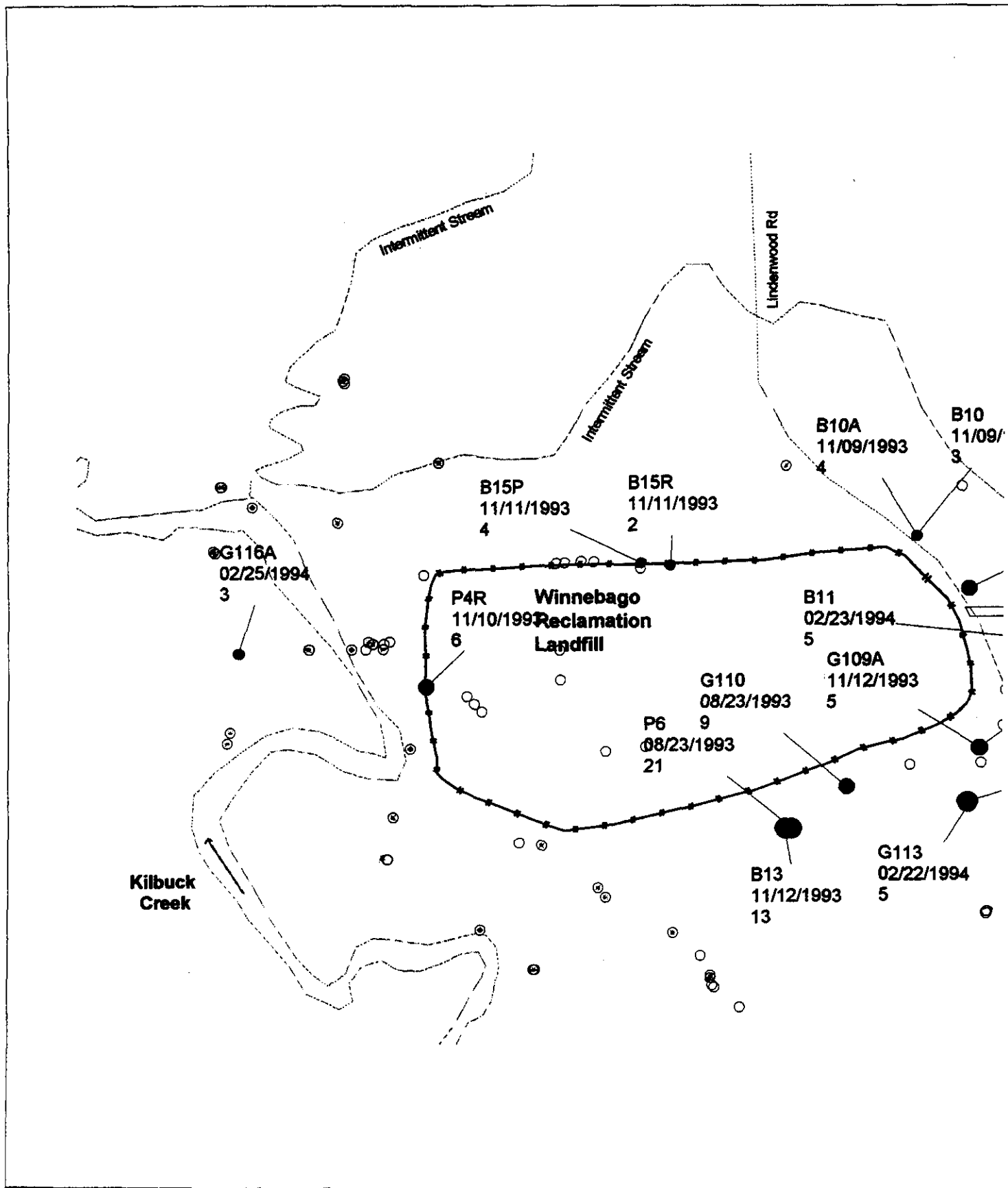
Checked by: PAR

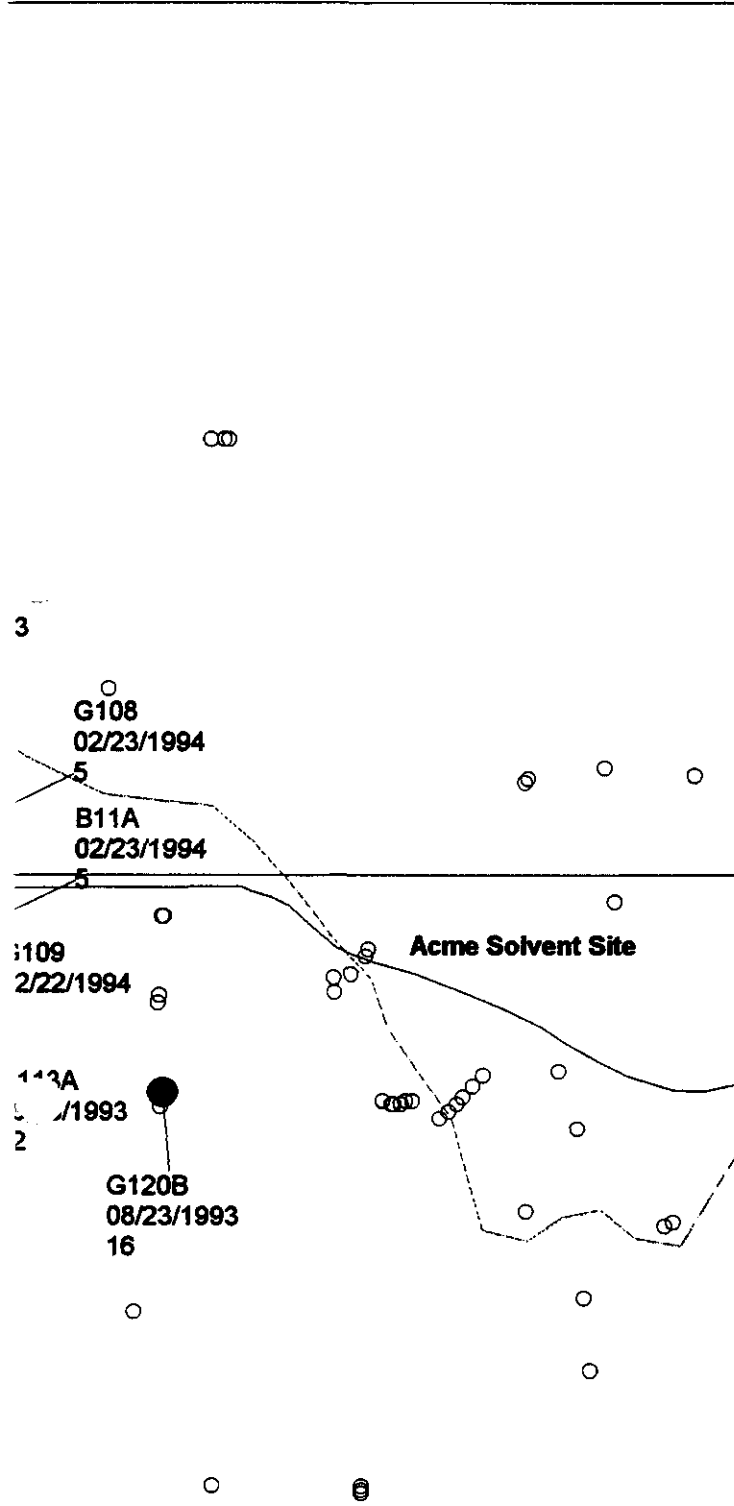
Revised: 6/21/95

Plotted by: MPM

Drawing: pce.wor

**Figure
4.1-4**





- TCE (1993-95)**
Concentrations in ug/L
- > 10 (9)
 - 5 to 10 (15)
 - 1 to 5 (21)
 - < 1 (0)
 - nd (110)

Note: Labels indicate date and value of maximum detection.

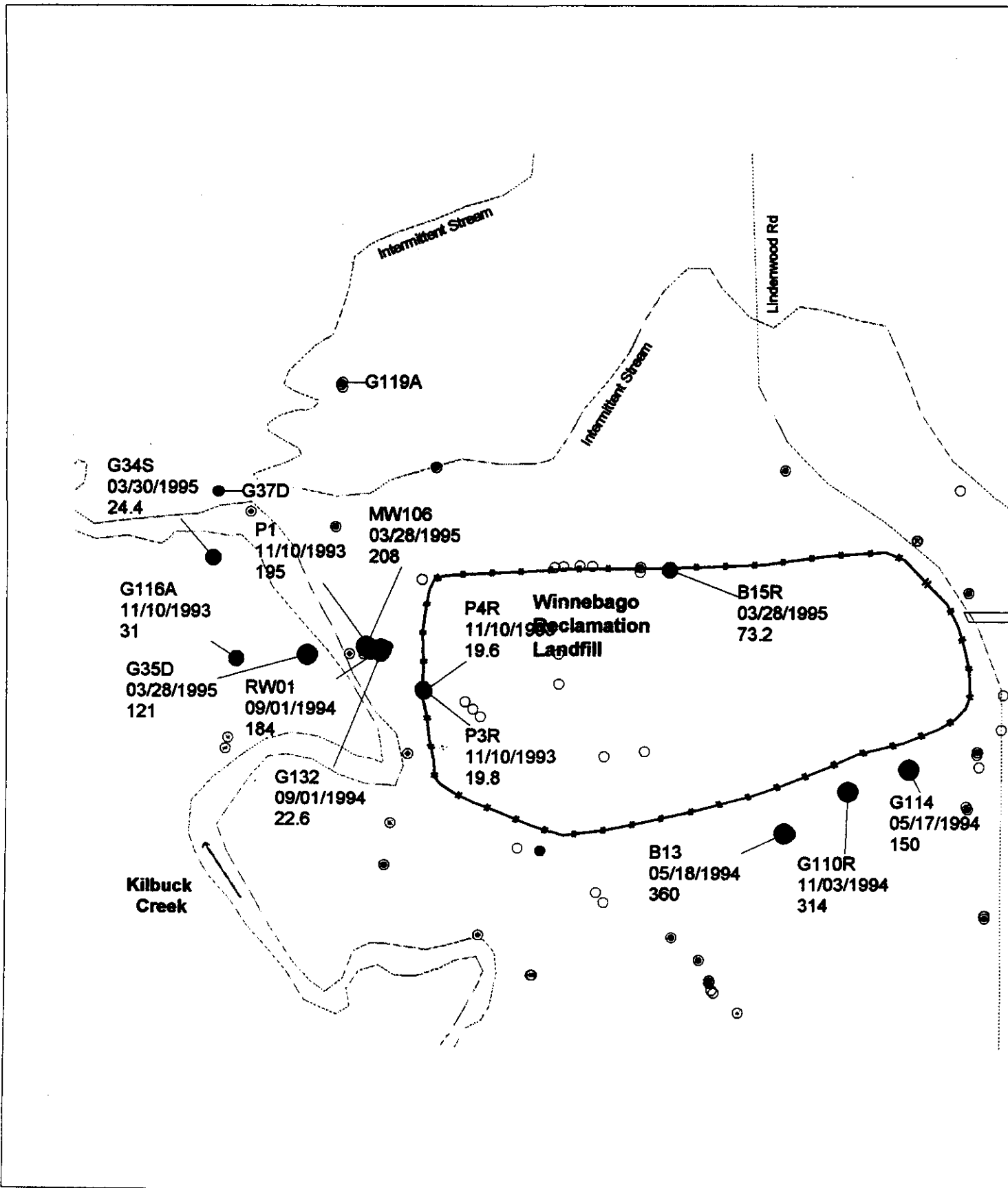


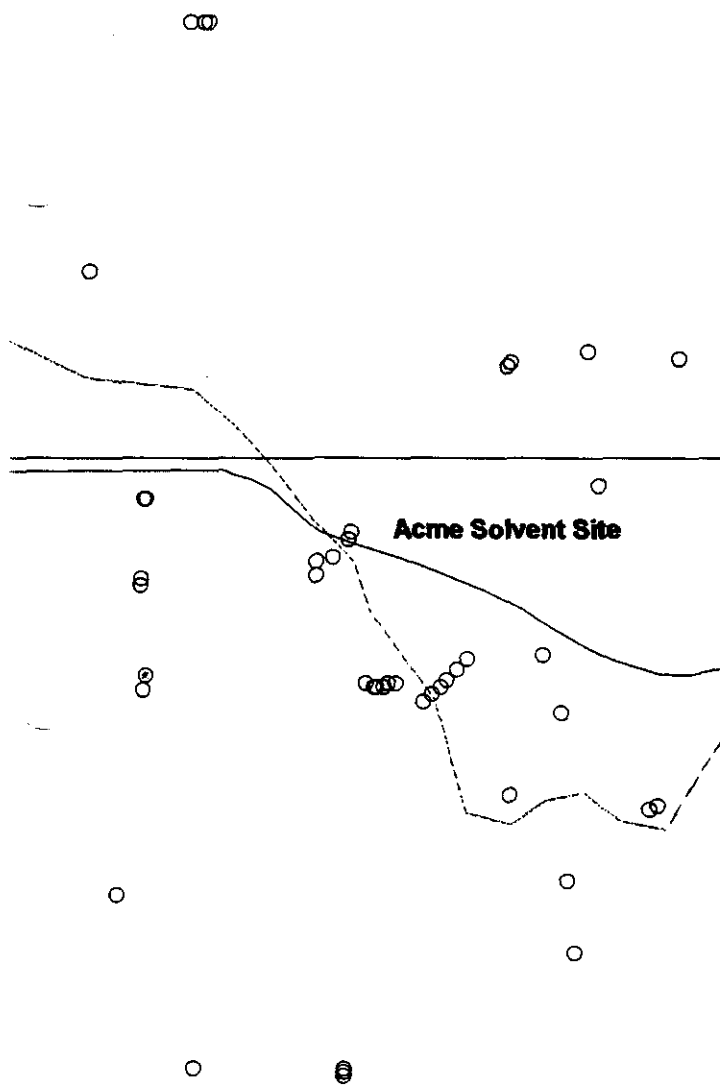
Concentration of TCE
at WRL Site Wells
(Current GeoTrans Database)



Prepared by: MPM	Date: 5/16/95
Checked by: PAR	Revised: 6/21/95
Plotted by: MPM	Drawing: tce.wor

Figure
4.1-5





Ammonia Nitrogen (1993-95)

Concentrations in mg/L

- > 100 (31)
- 10 to 100 (17)
- 1 to 10 (17)
- < 1 (60)
- nd (104)

Note: Labels indicate date and value of maximum detection.



Concentration of Ammonia Nitrogen
at WRL Site Wells
(Current GeoTrans Database)

GeoTrans, Inc.
ANALYTICAL SERVICES

Prepared by: MPM

Date: 6/21/95

Checked by: PAR

Revised:

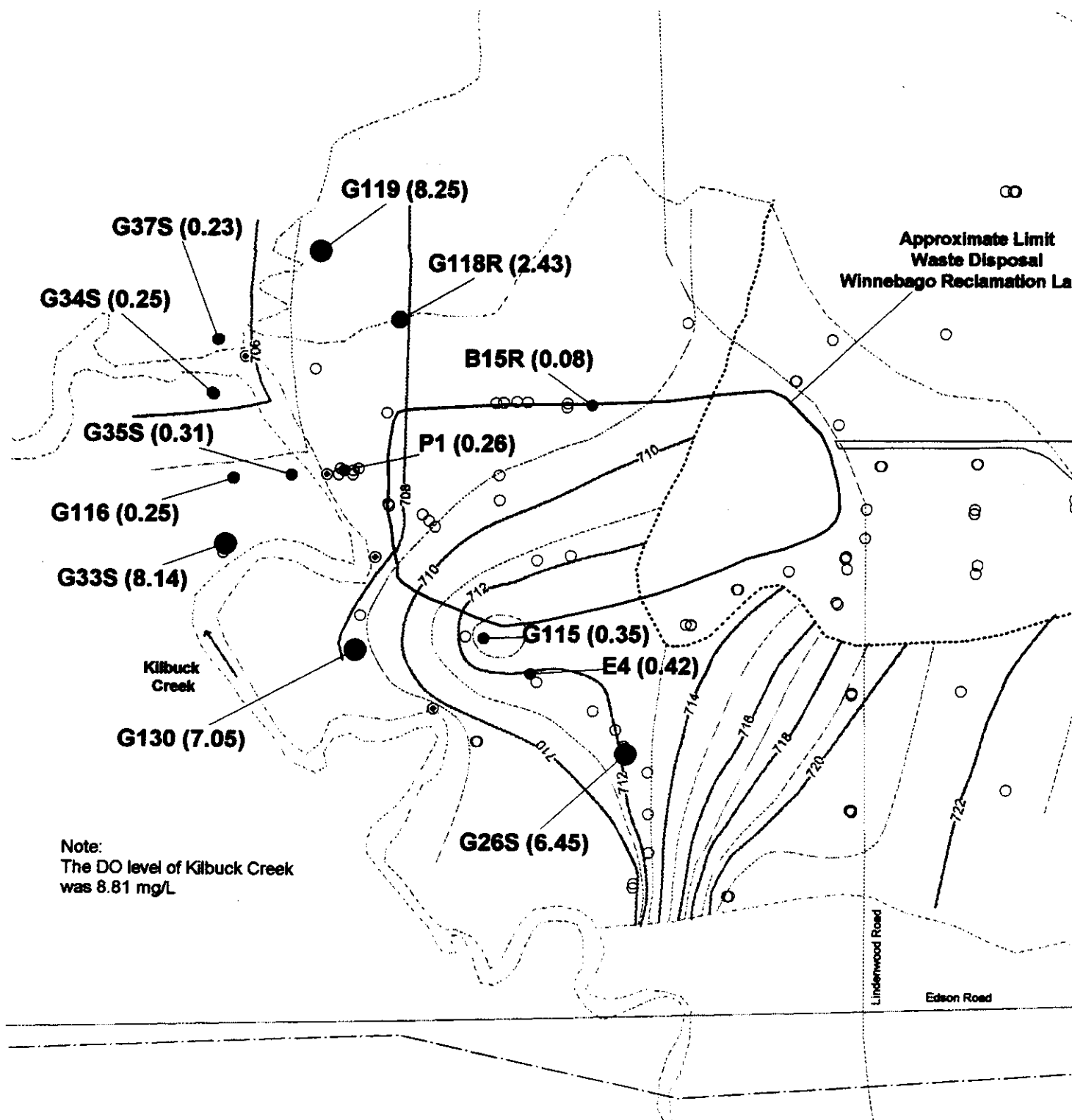
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Drawing: nh3_n.wor

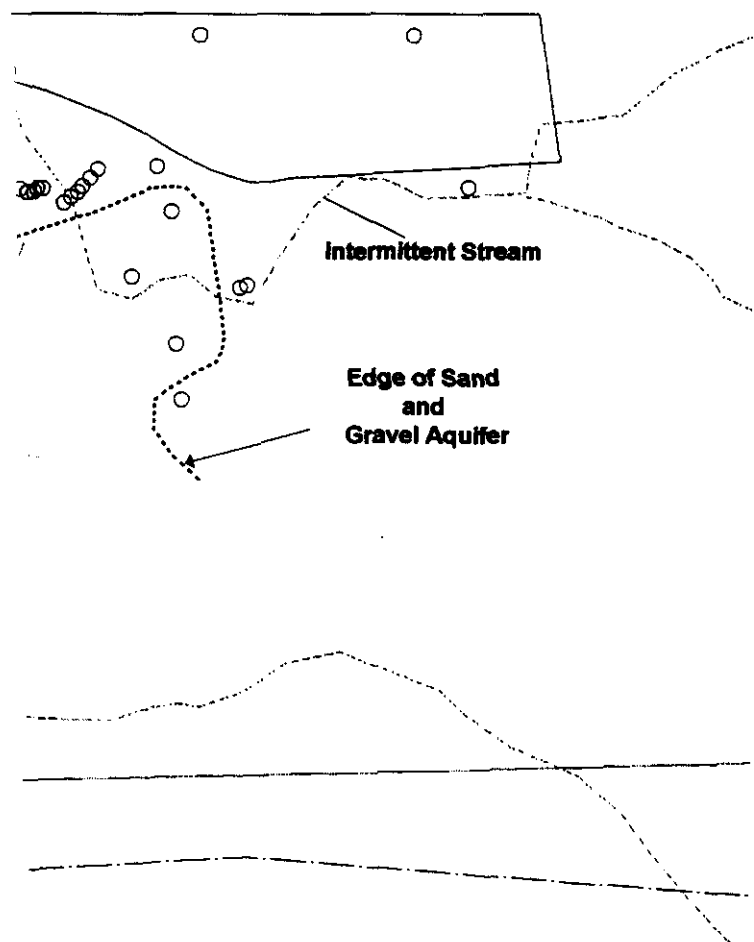
**Figure
4.1-6**

Table 4.1-1. Monitor well field parameter measurements.

Well Name	Water Depth	Well Depth	pH	DO (mg/L)	Specific Conductance (µmhos/cm)	Volume Purged (gal.)
B13	11.66	35.56	6.68	0.35	1226.00	2.50
B15P	31.27	65.22	7.48	0.88	572.00	6.50
B15R	10.85	47.70	6.93	0.08	2360.00	2.50
E2A	44.60	52.20	7.18	5.57	657.00	9.00
E3A	39.00	39.60	6.51	6.00	655.00	8.00
E4	12.25	20.00	6.58	0.42	1085.00	2.00
E4A	30.21	38.81	7.40	5.37	631.00	6.00
G26D	31.36	35.21	6.99	6.69	669.00	6.50
G26S	18.33	22.05	7.28	6.45	649.00	4.00
G33D	49.30	56.62	7.56	1.97	674.00	8.30
G33S	20.50	28.10	7.08	8.14	615.00	2.50
G34D	65.00	71.65	7.11	3.48	640.00	11.60
G34S	19.70	26.45	7.01	0.25	1045.00	2.50
G35D	49.70	56.70	7.09	0.26	1530.00	8.50
G35S	19.75	26.95	7.16	0.31	687.00	2.50
G36	35.35	40.00	7.28	5.90	656.00	7.50
G37D	78.25	82.20	7.33	0.26	706.00	15.65
G37S	12.05	17.55	7.23	0.23	685.00	2.50
G104	26.32	47.22	6.68	0.47	1417.00	5.50
G109	18.80	53.60	6.54	0.54	1078.00	4.00
G109A	45.90	81.60	6.53	0.11	1221.00	9.20
G111	17.30	38.50	7.10	5.70	641.00	3.50
G111A	36.60	57.60	7.17	5.30	651.00	7.50
G113	11.30	50.90	6.61	2.70	1002.00	6.00
G113A	36.65	77.90	6.48	0.40	1448.00	10.00
G115	15.70	22.95	6.03	0.35	1544.00	1.50
G116	15.15	22.05	7.15	0.25	624.00	1.64
G116A	45.10	52.12	6.95	0.30	1058.00	7.50
G116D	96.80	102.10	7.28	1.43	617.00	18.30
G118A	36.44	45.71	7.19	5.80	637.00	7.50
G118R	8.75	15.85	6.83	2.43	911.00	1.50
G119	7.70	--	7.09	8.25	668.00	3.00
G119A	12.20	--	7.20	6.23	680.00	7.70
G120B	117.50	150.00	7.13	3.50	578.00	23.50
G130	--	--	6.72	7.05	677.00	--
G130A	39.20	45.60	7.21	5.85	647.00	8.00
G131A	43.95	48.75	7.04	6.10	657.00	9.00
G132	81.00	100.95	7.52	5.15	570.00	16.50
MW106	43.24	61.49	7.20	0.36	4080.00	9.00
P1	17.79	37.99	6.98	0.26	2570.00	3.50
P4R	27.85	70.00	7.22	0.34	727.00	6.00
Kilbuck Creek	--	--	7.53	8.81	673.00	--



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Winnebago Reclamation Services

Dissolved Oxygen (DO) by Value Concentrations in mg/L

- > 5 (4)
- 1 to 5 (1)
- < 1 (8)
- × ND (0)

○ Existing Well

⊙ Existing Stream Gage

● G101 (160) Well I.D. and D.O. Concentration (mg/L)

— 750 Groundwater Elevation Contours (4/25/95)



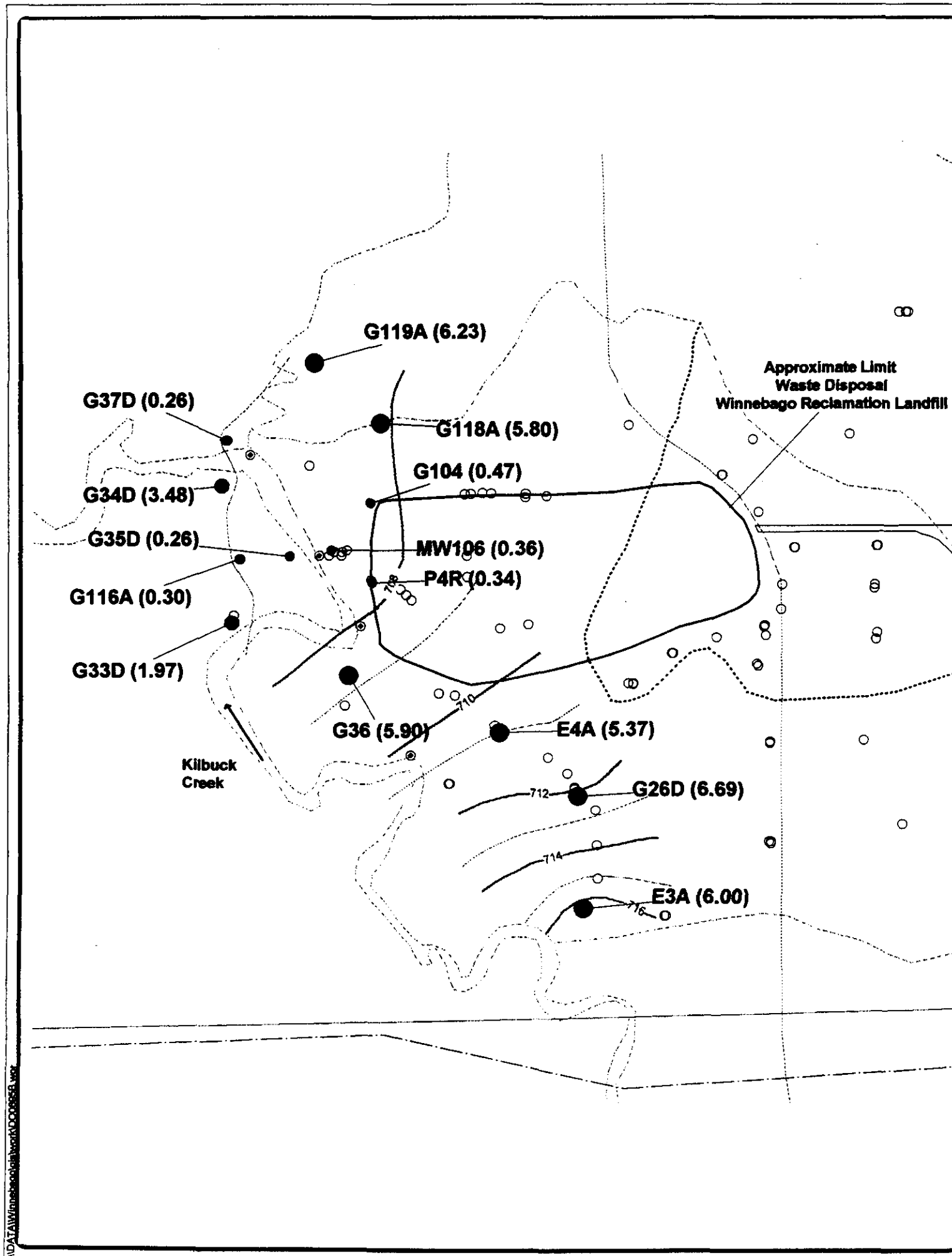
TITLE:
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in the Upper Zone of the Sand and Gravel Aquifer.

LOCATION:
Winnebago Reclamation Services, Rockford, IL.

GeoTrans, Inc.

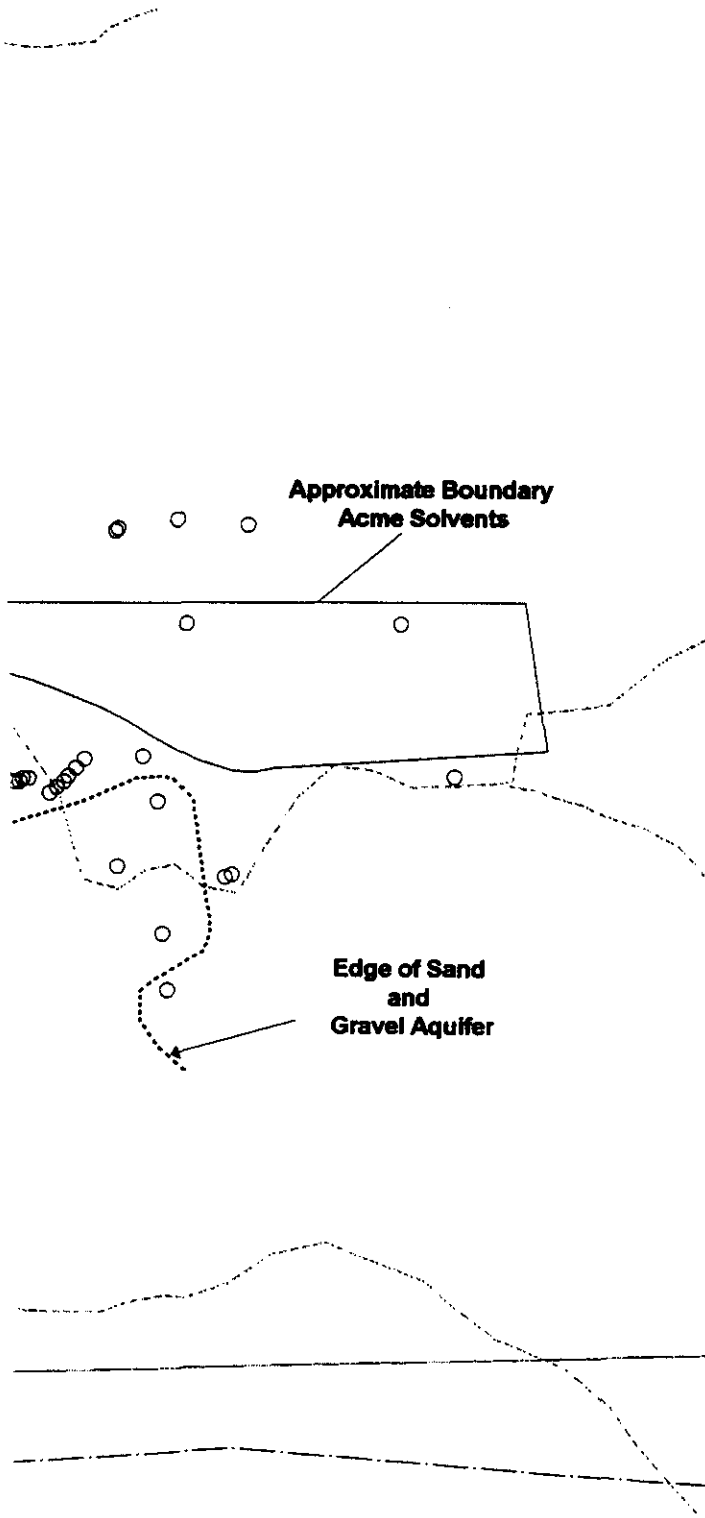
CHECKED:	D.B.
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FILE:	DO 0695.wor
DATE:	6-28-95

FIGURE:
4.1-7



7

7



Winnebago Reclamation Services

Dissolved Oxygen (DO) by Value Concentrations in mg/L

- > 5 (6)
- 1 to 5 (2)
- < 1 (8)
- × ND (0)



Existing Well



Existing Stream Gage

● **G101**
(160)

Well I.D. and D.O.
Concentration (mg/L)



750 Groundwater Elevation
Contours (4/25/95)

0 680 1360
feet

TITLE:

**Observed June 1995 Dissolved Oxygen Concentrations
in the Lower Zone of the Sand and Gravel Aquifer.**

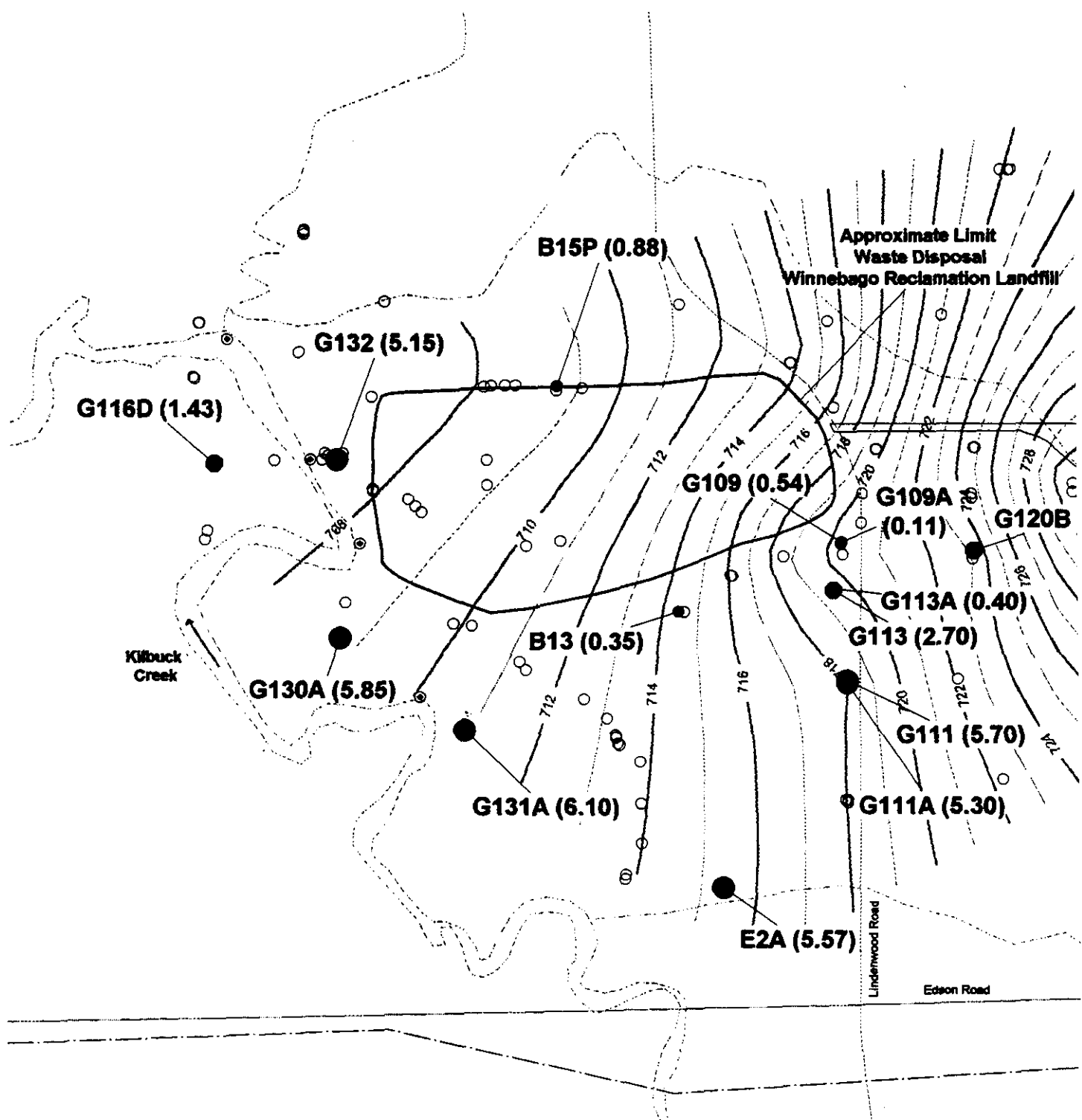
LOCATION:

Winnebago Reclamation Services, Rockford, IL.

GeoTrans, Inc.

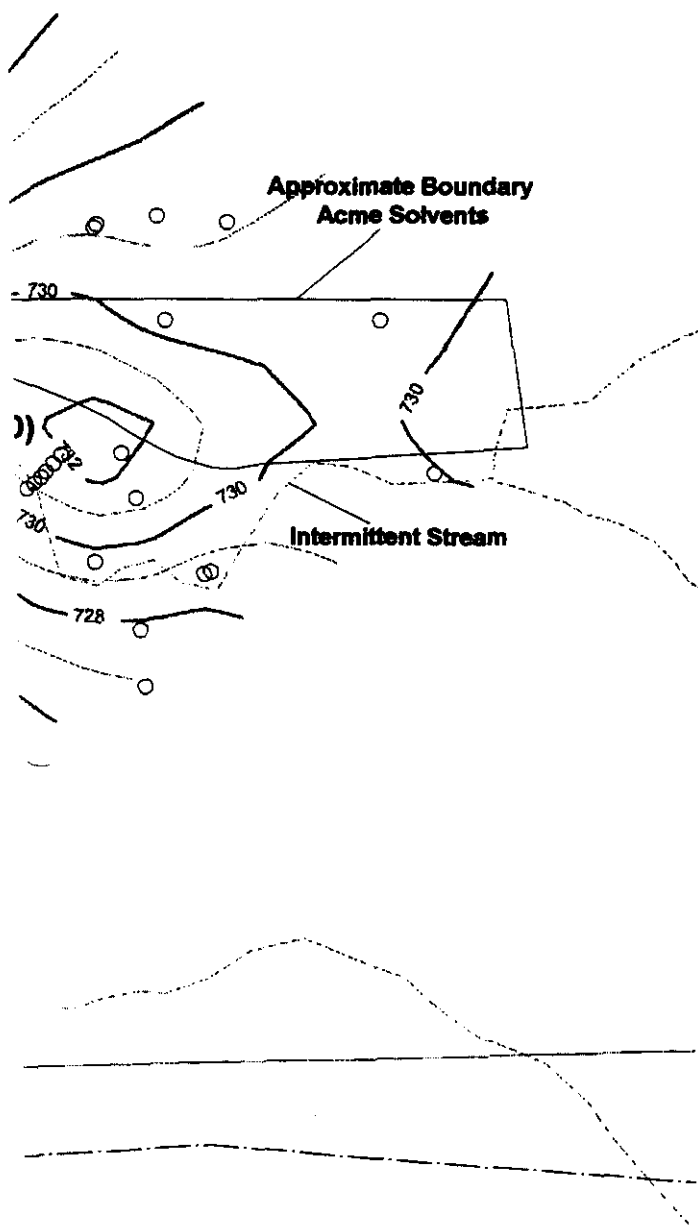
CHECKED:	D.B.
DRAFTED:	P.M.M.
FILE:	DO 695B.wor
DATE:	6-28-95

FIGURE:
4.1-8



(

(



Winnebago Reclamation Services

Dissolved Oxygen (DO) by Value
Concentrations in mg/L

- > 5 (6)
- 1 to 5 (3)
- < 1 (5)
- × ND (0)

○ Existing Well

⊙ Existing Stream Gage

● G101 (160) Well I.D. and D.O. Concentration (mg/L)

— 750 Groundwater Elevation Contours (4/25/95)

0 680 1360
feet

TITLE:
Observed June 1995 Dissolved Oxygen Concentrations
in the Bedrock Aquifer.

LOCATION:
Winnebago Reclamation Services, Rockford, IL.

GeoT
Trans, Inc.

CHECKED: D.B.
DRAFTED: P.M.M.
FILE: DO 695C.wor
DATE: 6-28-95

FIGURE:
4.1-9

concentrations of dissolved iron, manganese, and other redox-sensitive inorganics may be due to the increased solubility of the reduced form of the inorganic constituents. This indicates that an increase in the dissolved oxygen content of the groundwater will 1) reduce dissolved inorganic concentrations by a reduction in solubility, and 2) decrease organic concentrations through enhanced biodegradation.

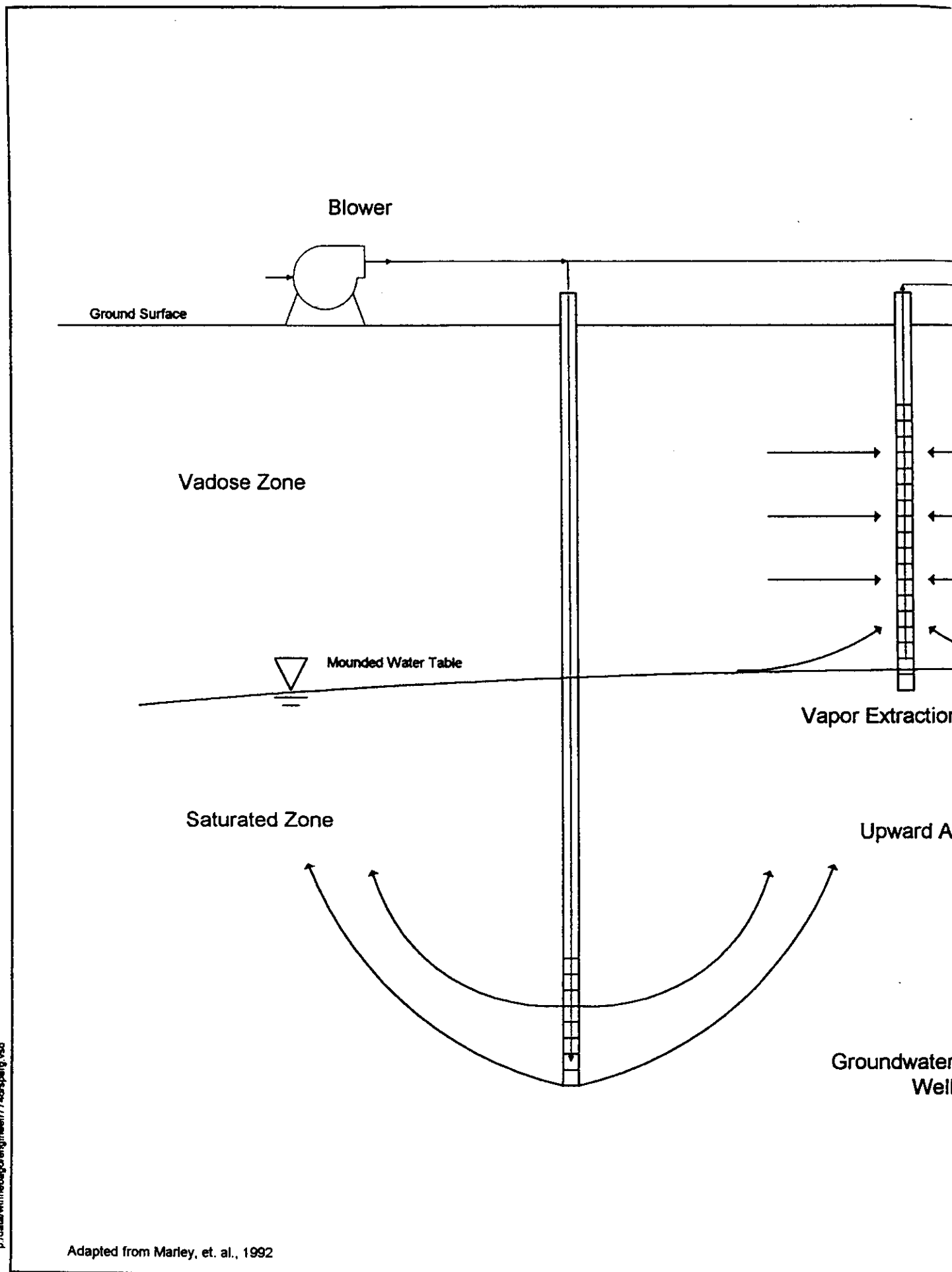
4.2 AIR SPARGING AND SOIL VAPOR EXTRACTION SYSTEM

Air sparging wells will be installed along the west side of the landfill to provide the *in-situ* treatment of groundwater that migrates from both the landfill and upgradient sources toward Kilbuck Creek. An SVE system is used to extract the contaminant vapor in the unsaturated zone in the vicinity of the sparging wells. Figure 4.2-1 is a schematic of a typical SVE/air sparging system, Figure 4.2-2 is a process flow diagram for the planned system, and Figure 4.2-3 shows the proposed locations of sparging and vapor extraction wells at the site. Operation of the SVE system may be halted at some point if extracted vapor maintains relatively low levels of VOCs. Startup test information will be used to estimate the mass of contaminant vapor that will be extracted from a full scale system to determine the effectiveness of SVE. If the full-scale SVE system is installed with the air sparging system it may be feasible to terminate its operation when extracted vapor concentrations reach a level agreed to by IEPA and USEPA.

4.2.1 INJECTION POINT RADIUS OF INFLUENCE

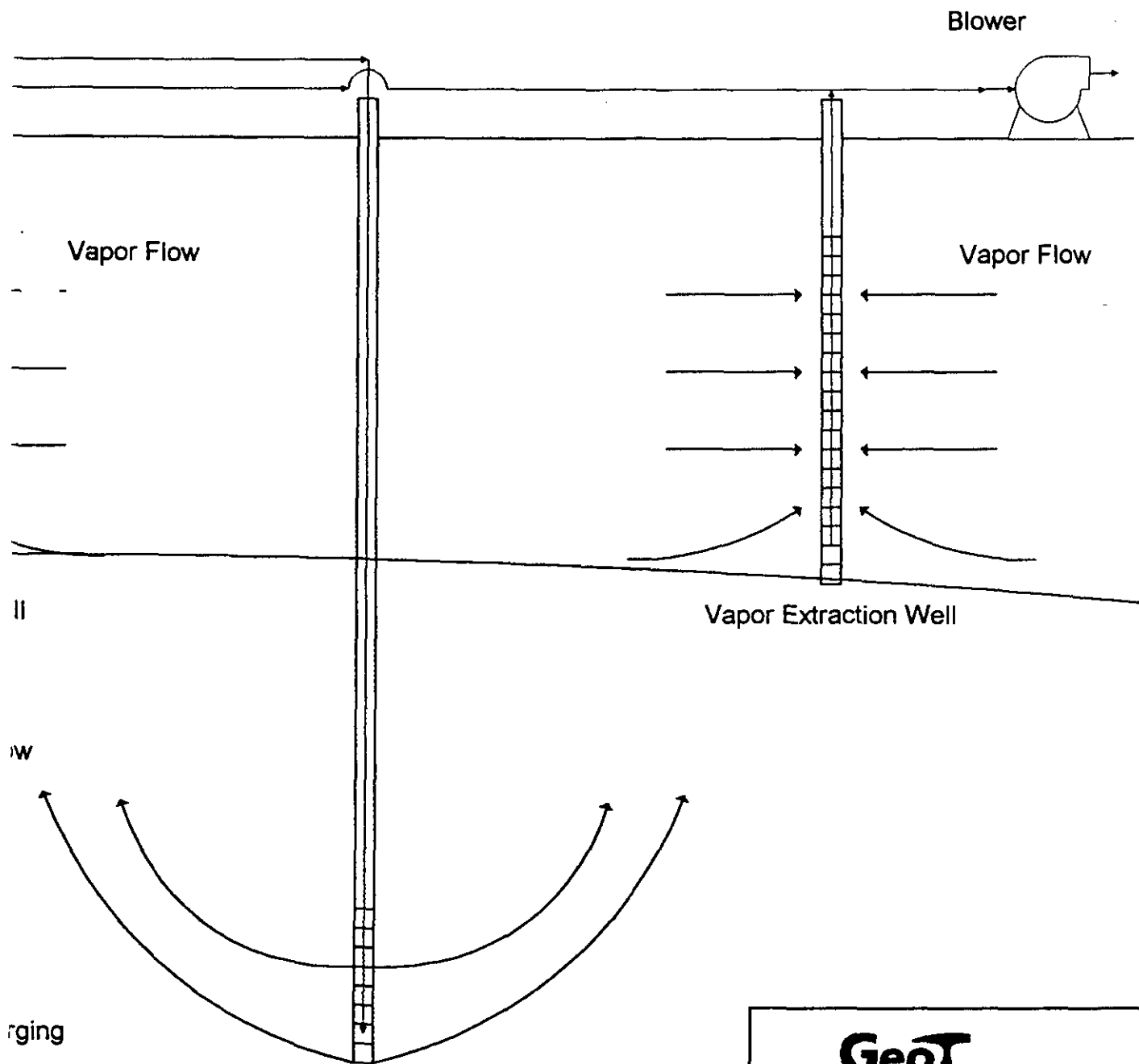
The injection point radius of influence is defined as the furthest distance traveled by air channels from the sparge point. In coarse soil formations, as at the WRL site, the radius of influence has been observed to be five to 20 feet (Marley et al., 1992).

It should not be assumed that contaminants within the radius of influence will be remediated at the same rate. In a heterogeneous saturated zone, the density of channels will be neither uniform nor radially symmetric about a sparge injection point. At some sites, confining strata may cause the air to migrate laterally from the sparge point, leaving



p:/data/winnibago/engineer/7740sparg.vsd

Adapted from Marley, et. al., 1992



GeoTrans, inc.
GROUNDWATER SPECIALISTS

Fig. 4.2-1 Schematic of Soil-Vapor Extraction and Air Sparging System

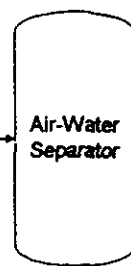
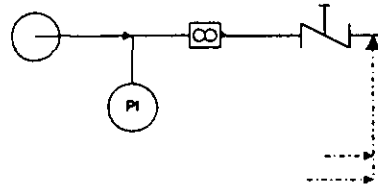
Prepared by: MPM

Date: 05/17/95

Checked by: PAR

Dwg: sparg.vsd

Vapor Extraction Well (1 of 13)



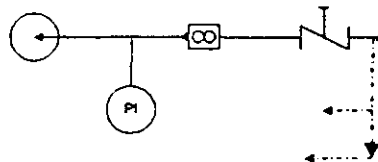
Flowmeter

PI

Vapor Extraction Blower

System Housing Unit

Air Sparging Well (1 of 31)



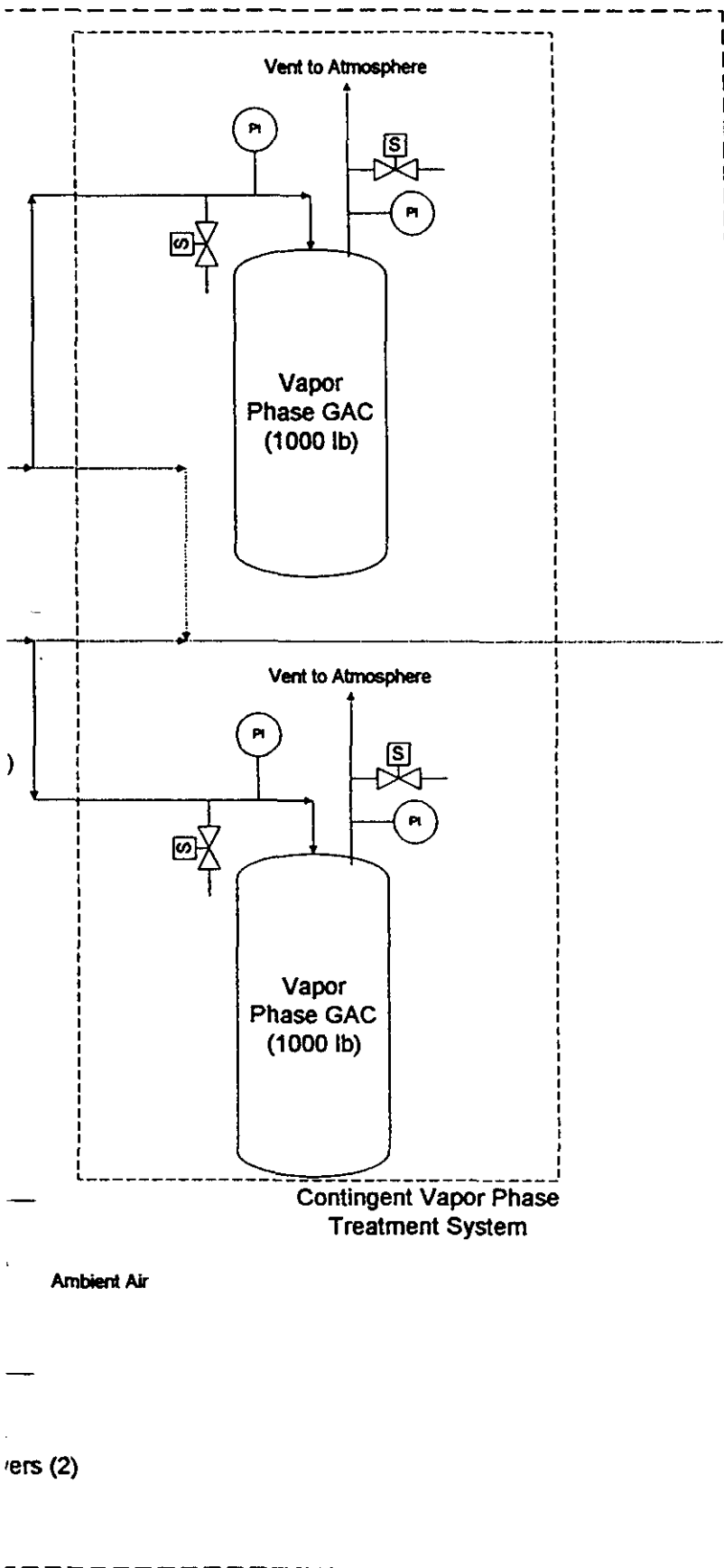
Flowmeter

PI



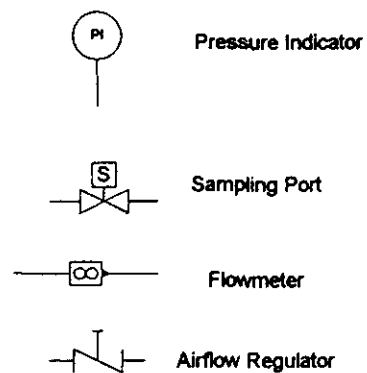
Air Sparging I





Contingency: Off-Gases to
NRG Technologies, Inc.

LEGEND



GeoTrans, inc.
GROUNDWATER SPECIALISTS

Figure 4.2-2 Process Flow Diagram for
Air Sparging / SVE System

Prepared by: MPM

Date: 05/17/95

Checked by: PAR

Dwg: asprsys.vsd

Housing for System Equipment
(Blowers, A/W Separator, etc)

System Piping

WRL Access

Air Sparging Wells
(31 wells at 20 ft)

Venting Wells
(13 wells at 50 ft)

Limit of Fill

Kilbuck Creek



Road

enters)

enters)

Winnebago Reclamation Landfill

N



LEGEND

✧ Air Sparging Well

⊕ Vapor Recovery Well

0 130 260



feet

Air Sparging System:
Well Locations and System Piping

GeoTrans, Inc.

Prepared by: MPM

Date: 5/16/95

Checked by: PAR

Revised:

Plotted by: MPM

Drawing: aspeys.wor

Figure
4.2-3

untouched volumes of water between the sparge injection point and the ultimate exit points of air channels (Ahlfeld et al., 1994). However, the WRL site geology is relatively homogeneous, so the radius of influence of injection points should be consistent and remediation relatively uniform.

4.2.2 AIR INJECTION WELL INSTALLATION

The number and placement of air sparging wells should be chosen to maximize air flow through the contaminated zone. Based on the predicted radius of influence, the number of air sparging wells is chosen to ensure that the contaminated zone is encompassed by the zone of influence of the collective system of individual wells. For the preliminary design process, the radius of influence is assumed to be ten feet (wells spaced at 20-foot intervals). This radius of influence estimate is empirically based; actual values will be measured in the field. Theoretical considerations indicate that increasing the number of wells will increase the rate of remediation. Installation of additional sparging/extraction wells will be a project contingency.

4.2.3 AIR INJECTION WELL DESIGN

The design of air sparging wells will include two elements: (1) the injection well screened interval; and (2) the depth of the screened interval with respect to the static groundwater level. The screened section of an air sparging well must be located within the saturated zone. Water table fluctuations must be considered, and the top of the well screen must be placed at a depth where it will not become exposed if the water table drops. A relatively permeable packing material, usually sand, will surround the well screen, and a bentonite seal and bentonite/cement grout will seal the annulus above the well screen to inhibit short-circuiting of the injected air. PVC is typically used for the injection well construction. Stainless steel is required if steam or hot air injection is selected. Injection well diameters typically range from one to four inches; performance is not expected to be affected significantly by increasing the well diameter (Johnson et al., 1993). Short screened intervals, on the order of one to three feet, are generally used in air sparging wells because

most of the air exits through the top of the screened interval, where the pressure head is a minimum (Marley et al., 1992). Use of longer screen intervals does not significantly add to the effectiveness of the process.

For the WRL site, 31 air injection wells with an average depth of 90 feet below ground surface (BGS) to the top of bedrock will be installed to serve as air sparging wells. These wells will be installed along the west side of the landfill to establish a 600-foot-long sparging zone for the treatment of groundwater migrating from the landfill vicinity. Figure 4.2-3 shows the location of the 31 wells. According to the cross section maps, water levels are estimated to be between 20 to 40 feet BGS. The saturated zone thickness is about 50 to 70 feet. Air sparging wells will be two inches in diameter with a two-foot screened interval. Material used for construction of the wells will be PVC pipe. An appropriately sized sand pack will be placed around the well screen to about one foot above the screen, then two feet of bentonite seal and bentonite/cement grout will seal the annulus to the surface. The exact depth of each well will be determined during installation. The decision will depend upon hydrogeologic information obtained during drilling.

4.2.4 AIR RELIEF WELLS

Soil vapor extraction is used in conjunction with air sparging systems to remove contaminant vapors liberated by the air sparging process and to minimize the potential migration of contaminant vapors. Soil vapor extraction wells for air sparging applications are constructed in the same manner as those used in traditional soil vapor extraction applications. Soil vapor extraction wells are typically screened from the capillary fringe to near the ground surface to capture the vapor from the sparging wells. It is important to consider groundwater level fluctuations when choosing the locations of the well screen.

Radius of influence for the sand and gravel formation is estimated at 50 feet (RACER, 1994). Thirteen air relief wells, spaced at conservative 50-foot intervals, will be installed to vent the contaminant vapor out of the vadose zone (Figure 4.2-3). The average depth of the soil venting wells will be 30 to 40 feet, depending on the thickness of the unsaturated zone.

4.2.5 INJECTION AND EXTRACTION AIR FLOW RATES

For the air sparging process, it is essential to provide a sufficient air-to-water ratio to produce the desired contaminant mass removal. The hydraulic control system should provide sufficient retention time for the groundwater that flows through the sparging points. Air flow rates typically used in successful air sparging applications are in the range of three to ten standard cubic feet per minute (scfm) per sparge point (Marley et al., 1992). Pulsing of the air flow into the air sparging wells should be considered due to potential mass transfer limitations between vapor and liquid phases. It also provides an energy-efficient and cost-effective approach to remediation. Detailed operation considerations are presented in Section 6. In order to make sure all injected air is recovered by the soil vapor extraction wells, the extraction air flow rate will be greater than the injected flow rate.

Two 50-hp blowers, which can supply up to 150 scfm of air each, are selected to supply up to ten scfm to each sparging well. The volume of air vapor to be extracted will be up to 560 scfm (two 5-hp blowers, supply pressure of two to three psig), 30 percent more capacity than the air injected into the saturated zone. This is to ensure the vapor that contains VOCs is not transferred offsite. Air will be delivered to the injection wells by steel or HDPE piping and air will be extracted through HDPE piping. Pressure and flow gauges and regulators will be supplied at each well to measure air flow rates.

4.2.6 INJECTION PRESSURE

Air injection pressures are governed by the static water head above the sparge point, the air entry pressure of the saturated soils, and the gas injection operating flow rate (Marley et al., 1992). The minimum operating pressure for the air injection blower is equal to the pressure head at the top of the well screen plus the air entry pressure required to overcome capillary forces. In the design process, the lowest effective air injection pressure will correspond to the pressure required to maintain the minimum gas flow rate that will achieve the desired stripping efficiency. Higher flow rates may be necessary to provide a more uniform gas channeling distribution in heterogeneous soils, but they can also cause the formation of significant subsurface gas pockets due to bubble coalescing (Marley, 1992).

High air pressure may also create fractures in the sparging well annular seal or along weak joints in the soil resulting in a loss of system efficiency.

According to previous air sparging field data, the applicable air injection pressures ranged from one to eight psig for shallow groundwater aquifers (Johnson, 1993). For the WRL site, air is to be injected into the saturated zone 40 feet below the water table, therefore, at least 22 psig of air pressure is needed. The blowers selected for this design will be able to deliver up to 30 psig of air pressure to the sparging wells.

4.3 CONTINGENCY

The following section describes contingent operations and processes which may enhance and optimize system operation and/or compliance with ARARs. Contingent processes include off-gas treatment, steam injection, and enhanced oxygen supply.

4.3.1 OFF-GAS TREATMENT

Discharge of extracted vapors must be in compliance with State air discharge standards. This may require the use of off-gas treatment such as vapor phase carbon beds, thermal oxidizers, or chemical oxidizers.

The highest total concentration of VOCs in site groundwater is less than 100 µg/L. Assuming the maximum 100 µg/L VOCs are removed by the air sparging and soil vapor extraction system, the VOCs loading in the vapor stream would be less than 4.49 lb/d based on 500 cfm of extracted air. The standard for the emission of VOCs is eight lbs/hr (35 IAC 215.301). Vapor treatment may be required due to individual VOC emissions or the complete site management plan. If necessary, vapor phase carbon would provide economical removal of PCE and TCE, and will be the likely vapor treatment unless vinyl chloride levels are elevated. The vapor phase carbon units will be able treat up to 1000 cfm of air from the soil vapor extraction system. The SVE system is also expected to extract methane from the landfill. Estimated content of methane in the extracted vapor is about 50 percent. If acceptable for existing operations, extracted vapor may be sent to NRG facilities for

reclamation. Due to the expected methane concentrations, all extraction equipment and appurtenances will be explosion proof.

4.3.2 HEATED AIR/STEAM INJECTION

Heated air can be used to heat soils and increase degradation and volatilization rates. However, at feasible air flow rates and temperature differences, it is not possible to significantly warm soils and groundwater to increase effectiveness significantly (Johnson et al., 1993). Steam injection is possible, but would add extreme cost requirements unless a steam source is readily available.

4.3.3 ENHANCED OXYGEN SUPPLY

Air sparging depends on two basic process for contaminant removal: volatilization and aerobic biodegradation. Air sparging is one of a number of methods for delivering oxygen to the saturated zone, and, therefore, has the potential to stimulate aerobic biodegradation. Conventional *in-situ* oxygen delivery processes have either used the injection of oxygenated water or water containing hydrogen peroxide to increase subsurface oxygen levels. A new enhancement to speed up biodegradation involves the usage of ozone. Meticulous application of ozone in air sparging wells can remove recalcitrant organic compounds which are often resistant to more traditional approaches such as aerobic bioremediation and volatilization (Nelson, 1995). At present, the best commercial candidates for ozonation are sites with chlorinated solvents, polycyclic aromatic hydrocarbon (PAHs) and PCBs. Another innovative technology available to deliver oxygen into the aquifer is by using Oxygen Release Compounds (ORC). ORC filter socks can be installed into air sparging wells and oxygen will be released at a slow, controlled rate when hydrated (Griffiths, 1995). Additional wells may be required with this option to provide adequate oxygen production. ORC or ozone application can eventually replace the air sparging system at the WRL site if aerobic biodegradation effects are determined to control the groundwater remediation without volatilization.

4.4 STARTUP TESTING

Air sparging requires a balanced airflow to maintain effectiveness and control. Design data determined in tests include the radius of influence of the air sparging system at different injection flows or pressures, the radius of influence of the soil vapor extraction system, and the pressure and vacuum requirements for effective capture and treatment of volatilized materials.

A number of different parameters can be measured during the tests to determine radius of influence such as VOC concentration in the soil and groundwater; VOC type; oxygen and carbon dioxide levels in the soil vapor; and water level increases in the wells. Using multiple parameters allows for cross correlation during design to determine effective airflow through the area of contamination and ensure the capture of the volatilized materials.

4.5 COST ANALYSIS

Tables 4.5-1 and 4.5-2 present a cost estimate of the conservative preliminary design. A startup period of four weeks is estimated, and the operation and maintenance period is estimated to be 10 years, as the clean-up time is projected to be ten years. As indicated in Tables 4.5-1 and 4.5-2, total capital cost is \$421,445 and annual operation and maintenance cost is \$36,800. The total present worth of this remediation project \$284,155 is for 10 years at a five percent discount rate. The cost estimate was generated using the Remedial Action Cost Engineering and Requirements (RACER) system developed by the U.S. Air Force Center for Environmental Excellence (AFCEE).

4.6 SCHEDULE

The remedial action will include three operational phases. In the first stage, two air sparging wells, a soil vapor extraction well and several monitoring wells will be installed in approximately one month. A test with these wells will determine the final design parameters. This startup test is expected to take two to three weeks. A final design will then be completed in approximately three months, and submitted for review by the appropriate

regulatory agencies. Pending a timely acceptance of the final design, the system components and configuration will then be specified, a contractor will be selected, and the system installed. These steps, including building and electrical permits, will require about four to six months. Another one month will be used to startup the air sparging system. The system should be planned to be fully operational in the summer of 1996 as indicated in the schedule included as Table 4.5-3. Five years after full operation, a critical CERCLA design review will be conducted (see Section 5).

The total time to attain groundwater clean-up goals is expected to be ten years. Based on mass balance calculations, the estimated time to reduce the leachate head to two feet is approximately six years once full-scale operation of the leachate collection system commences. In the SIG MOD application, solute transport modeling was performed to evaluate the suitability of the landfill design for the period of time after clean-up of existing impacted groundwater. The transport modeling indicated that the combination of leachate head reduction to two feet and reduced infiltration through the final geomembrane cover would cause groundwater concentrations to decrease to below background levels within 100 feet of the landfill. The rate of clean-up should increase once the complete landfill cover is installed in approximately five years.

Table 4.5-1. Capital cost estimate: air sparging at Winnebago Reclamation Landfill Site.

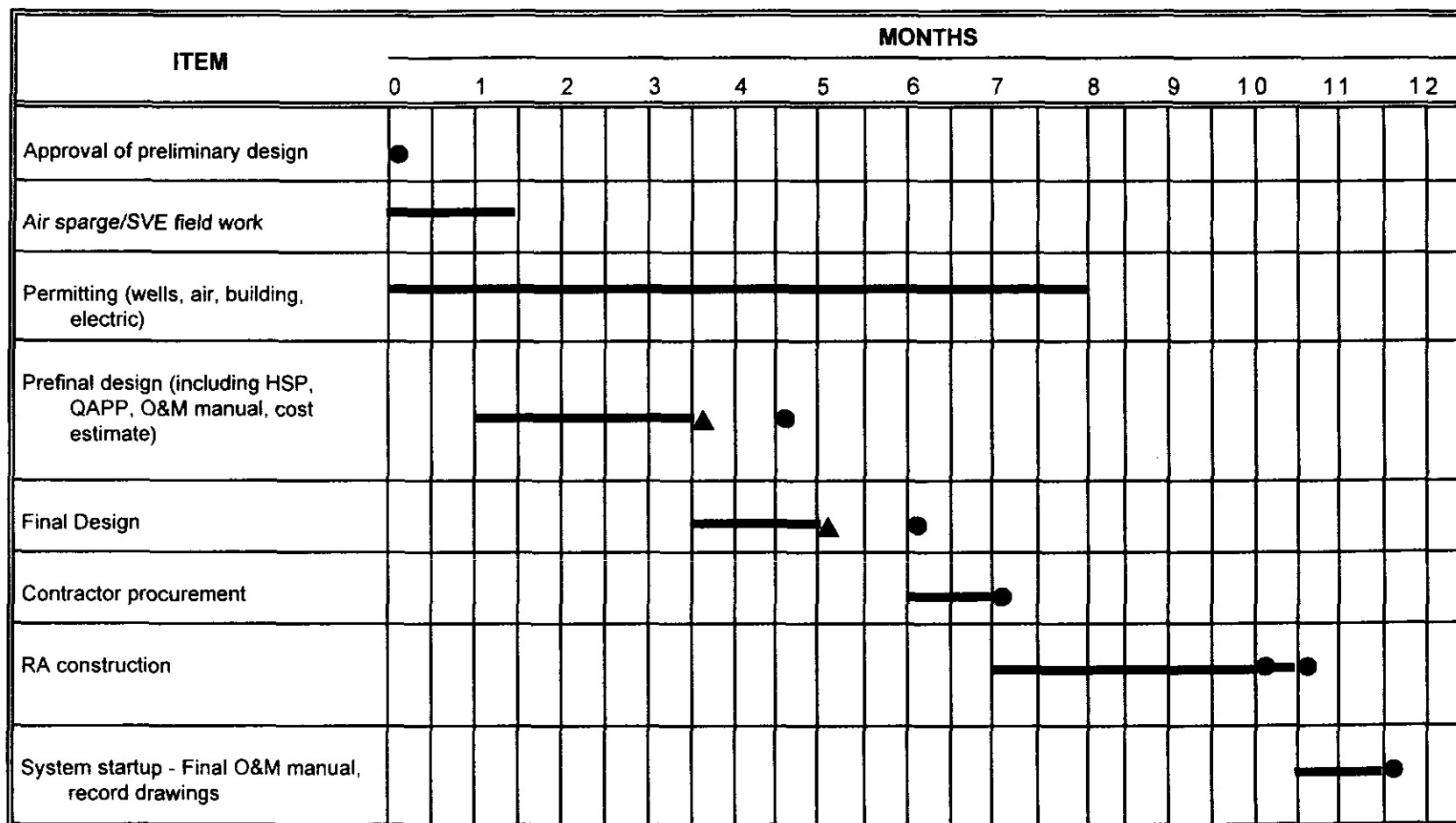
Description of Item	Quantity	Units	Unit Cost	Total Cost
Air Sparging				
Blower (163 SCFM, 30 psi, 15 HP)	2	EA	\$10,000	\$ 20,000
Piping	1,500	FT	\$15	\$ 22,500
Well Installation				
Air sparging well (90 feet)	31	EA	\$6,300	\$195,300
Well heads	31	EA	\$1,500	\$ 46,500
System Building				
Building/Foundation	300	SF	\$40	\$ 12,000
Electric/Piping and Valves		LS		\$ 8,000
Installation of equipment		LS		\$ 10,000
CAPITAL SUBTOTAL				\$314,300
Construction service ¹	30	DAY	\$1,000	\$ 30,000
Startup	15	DAY	\$2,000	\$ 30,000
Bid and scope contingency (15%)				\$ 47,145
TOTAL CONSTRUCTION COST				\$421,445

¹Well installation construction services included with installation costs.

Table 4.5-2. Annual O&M cost estimate: air sparging at Winnebago Reclamation Landfill Site.

Description of Item	Quantity	Units	Unit Cost	Total Cost
Electrical Power (40 HP) @ \$0.04/kwh	365	DAY	\$40	\$ 14,600
Inspection/Labor	12	MONTHS	\$500	\$ 6,000
Maintenance (6% of Capital Subtotal)		LS	\$16,200	\$ 16,200
TOTAL ANNUAL O&M				\$ 36,800

Table 4.5-3. Schedule for air sparging implementation at WRL site.



Note: CERCLA design review to occur at month 72.

- Document acceptance or meeting.
- ▲ Submittal for IEPA/EPA review (30-day review assumed)

5 MONITORING OF SYSTEM EFFECTIVENESS

Monitoring will be carried out to assess the performance of air sparging operating conditions in accordance with 35 IAC 811.326. This will help determine if system adjustments or expansions are necessary, and also determine if offsite migration of contaminant vapors and contaminated groundwater is occurring. Monitoring activities will also assist in evaluating the effectiveness of natural biodegradation in the impacted zones downgradient of the landfill. System monitoring will include soil gas concentrations, induced vacuum levels, groundwater level elevation, contaminant levels in ground water, DO levels in the aquifer, and, if necessary, extracted vapor concentrations. *In-situ* response data are often subjected to a wide range of interpretations concerning validity and meaning (Johnson et al., 1993). Selected monitoring wells at various intervals downgradient from the sparging system will be sampled on a quarterly basis to determine aquifer conditions. Monitoring parameters should include concentrations of VOCs, DO, ammonia, chloride and other parameters of concern (Johnson et al., 1995). Existing wells will be used for this purpose to the greatest extent possible.

5.1 SYSTEM EFFECTIVENESS

System effectiveness will also be evaluated in detail during the system operation and at the five-year CERCLA review. Monitoring data accumulated during the first five years of system operation will be analyzed to determine the progress toward clean-up goals. The data should show significant decreases in concentrations of the major contaminants (VOCs and ammonia) in downgradient wells. The data should also show that contaminant mass reduction is attributable to air sparging, intrinsic biodegradation, and elimination of the source areas by the geomembrane cover and leachate-collection system. The air sparging system will also undergo a preliminary review after one year of full operation and detailed review after two years of full operation. In these reviews, operating parameters will be compared to design parameters. Deviations between operating and design parameters will be

corrected if the changes enhance system performance or cost-effectiveness. Any necessary retrofitting and repair of the air sparging system components will occur to provide a more cost-effective system and to expedite the attainment of clean-up goals. Actual capital and O&M costs will be compared to those initially projected. This comparison will yield a better estimate of future operating and maintenance costs. During the detailed design review, statistical analyses will be performed on newly collected constituent data in both the upgradient background wells and the downgradient wells, to determine whether significant reductions are occurring. These analyses will also be performed to determine whether applicable groundwater quality standards are met. The results of these design reviews will provide a better estimate of the time to reach clean-up goals at the site. Remedy completion will be evaluated and demonstrated in accordance with 35 IAC 811.326 (e-g).

5.2 CONTINGENCY

If the design review, or significant data prior to the design review, indicates that the air sparging system is not effective in meeting clean-up goals, WRL will proceed with a contingent plan for groundwater migration control and treatment. As indicated in Section 3.2, FBR-GAC treatment could be used to remove VOCs and ammonia from the approximately 450 gallons per minute of groundwater extracted for required hydraulic containment goals. Pilot testing of the FBR-GAC equipment would be used to verify system effectiveness. Design, monitoring, and reviews for the groundwater recovery and ex-situ treatment system would proceed similarly to that described herein for the air sparging system. If information developed after implementation of the remedy has begun, it is determined that compliance with requirements of Section 811.325 (b) are not being achieved through the selected remedy, an alternative method will be evaluated in accordance with 35 IAC 811.326 (b).

6 SYSTEM OPERATION AND MAINTENANCE

Air injection equipment must produce a contaminant free vapor to avoid introducing new contaminants to the aquifer. Additionally, the air sparging/soil vapor extraction systems will be constructed in such a manner that air injection ceases automatically whenever the SVE system malfunctions, if the SVE system is being used to remove VOCs.

There are at least three distinct approaches to operating air sparging systems (Johnson, 1993). There can be staged, continuous, or pulsed operating strategies. In the staged approach, the unsaturated soil zone is remediated first, followed by air sparging. Continuous and pulsed systems are differentiated by continuous or intermittent air injection. If mass transfer limitations prove to govern air sparging system behavior, continuous operation will probably be the preferred option. Should the pulsing of the air injection enhance air/water mixing in the subsurface, a properly timed pulse operation could deliver enhanced performance.

Air sparging theoretically decreases the aquifer conductivity due to the decrease of the soil wetability, thus producing a lower permeability barrier to natural downgradient water flow. Such a lower permeability barrier could in turn produce an increase in head upgradient of the sparge zone and a tendency for some water to flow around the sparge zone. This phenomenon is expected to be of minimal concern at the WRL site due to the coarse-grained soils in the saturated zone. Another concern with a sparge interception system is that some of the water that migrates through the sparging zone would not come sufficiently close to air channels for contaminant removal to occur. To prevent those problems, air sparging can be operated in a cycling fashion with the air injection varied over a range of pressure or simply turned on and off with some frequency. The cycling will likely induce agitation and mixing of the water as the air channels form and collapse during each cycle (Ahlfeld et al., 1994). The WRL system will be designed to accomplish this pulsing effect, if desired.

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APPENDIX
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A Conceptual Model of Field Behavior of Air Sparging and Its Implications for Application

by David P. Ahlfeld, Amine Dahmani, and Wei Ji

Abstract

The basic physics of air flow through saturated porous media are reviewed and implications are drawn for the practical application of air sparging. A conceptual model of the detailed behavior of an air sparging system is constructed using elements of multiphase flow theory and the results of recent experimental work. Implications of the conceptual model on air sparging topics are discussed. The meaning of radius of influence in the context of air sparging is found to be ambiguous. The hydrodynamic effects of air sparging such as mounding of ground water and flow impedance are explored. Limitations on rates of remediation and operational strategies for improving sparging effectiveness are examined.

Introduction

Air sparging has recently gained significant attention for the remediation of volatile contaminants in the saturated zone. A typical system has one or more subsurface points through which air is injected. The injected air migrates toward the unsaturated zone, volatilizing contaminants from the ground water and delivering oxygen to the ground water as it migrates. The contaminated vapors are typically extracted from the unsaturated zone with a soil vapor extraction (SVE) system. Techniques for its application, advantages, and potential pitfalls have been described by Marley et al. (1992) and Johnson et al. (1993).

Air sparging has been used since the mid-1980s in Germany where the removal of chlorinated solvents was enhanced by injecting air into the saturated zone (Bohler et al. 1990). More recently, air sparging has been used in the United States for the cleanup of saturated zones contaminated with gasoline and other volatile contaminants. Numerous accounts describing application of the technique have been presented in the literature and are summarized in the review of Johnson et al. (1993).

In this paper we review some of the common assumptions that have been presented in the literature and appear to be prevalent in practice about the behavior of air sparging in light of the broader theoretical and experimental understanding of the flow of air in media which is initially water saturated. We begin by

describing elements of the theory underlying air sparging behavior: the simultaneous flow of two fluids (air and water) in a porous media. Combined with recent experimental work this yields a clearer conceptual model of air flow in typical air sparging applications. Using this conceptual model we provide our view of the implications of the theory for application of the method.

A Conceptual Model of Air Sparging Behavior

Air and water present in a porous media exhibit complex flow behavior; this behavior depends on both media properties and properties of the individual fluids. The effectiveness of air sparging depends on the volume of the aquifer reached by the air and the amount of mass transfer between the gaseous and aqueous phases. A fundamental problem in analyzing the behavior of an air sparging system is understanding the pathways taken by the air. The flow of two or more fluid phases in a porous media has been studied for many years by hydrogeologists, soil scientists, and petroleum engineers (Bear 1972; Dullien 1992). A solid theoretical framework has been established and extensive experimental results have been produced. We briefly review some of the results that will have implications for the behavior of air sparging systems.

Multiphase Flow Theory Relevant to Air Sparging

Under a wide range of conditions, flow of a single fluid in a porous media is governed by Darcy's law (Bear 1972), which takes the form, in one dimension:

$$q = -k \rho g / \mu \partial h / \partial x \quad (1)$$

where q is the fluid flux, k is intrinsic permeability, ρ is density, g is the gravitational constant, μ is viscosity, and h is the fluid head, given as the sum of pressure head and elevation head. In the present analysis Darcy's law will be used to describe the flow of water in the presence of air. Note that density and viscosity are properties of the fluid alone. For the case of flow through the media of a single fluid phase (e.g., water saturated porous media), k depends on various properties of the solid soil matrix. While no generally accepted expression for k exists, empirical relationships often express it in terms of the porosity, grain size distribution, grain packing, and tortuosity of the media.

In the case when two fluids are present in the media, the coefficients on the head derivative in Darcy's law are different for each fluid. For fluid i , Darcy's law is written as:

$$q_i = k k_{ri} \rho_i g / \mu_i \partial h_i / \partial x \quad (2)$$

where the i subscripts denote the respective property for fluid i . Here, k still depends only on the solid matrix, but it is now multiplied by a factor, k_{ri} , the relative permeability for fluid i , which varies from 0 to 1.0 and is different for each fluid. The relative permeability depends on the quantity of each of the fluids present.

Figure 1 displays the typical behavior of relative permeability of water for a sandy soil as a function of water saturation (Bruce 1972; Van Genuchten 1980; Dullien 1992). At 100 percent water saturation, the relative permeability is 1 and Equation 2 reduces to Equation 1. As water saturation decreases so does the relative permeability. Depending on the medium, at values between about 5 and 40 percent water saturation, the relative permeability essentially reduces to zero. At this point capillary forces, which hold the water in small pore spaces, dominate and water flow essentially stops, even though water is still present in the media.

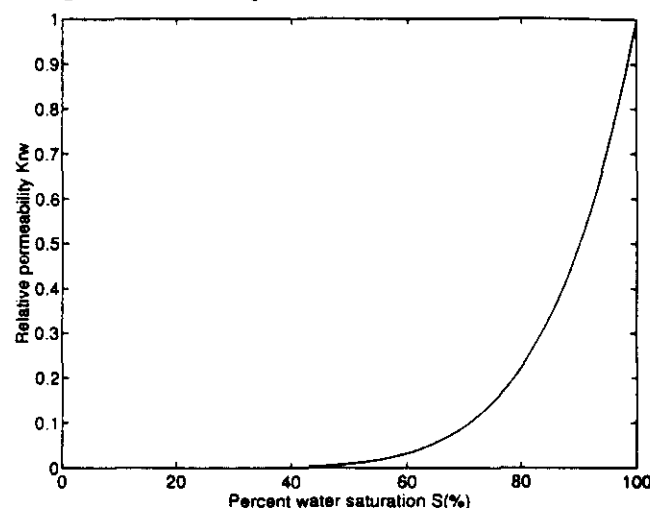


Figure 1. Relative permeability as a function of water saturation for a typical sandy soil.

In a two-fluid system in porous media, the two fluids will have many points of contact. At these interfaces a pressure exists in each fluid. However, under steady-state conditions, the pressure of the two fluids at the interface is not equal. The surface tension that exists between the two fluids at the interface produces a pressure difference which is defined as the capillary pressure — the difference between the two fluid pressures:

$$p_c = p_a - p_w \quad (3)$$

where p_c is the capillary pressure, p_a is the air pressure, and p_w is the water pressure. This capillary pressure can be described, under idealized conditions, by the Laplace equation:

$$p_c = 2 \sigma / r \quad (4)$$

where σ is the surface tension between the two fluids and r is the mean radius of curvature of the interface between the fluids (Bear 1972). Examination of this relation reveals that as r decreases the capillary pressure increases. Generally, r will decrease as grain size decreases.

Simulation models for air flow in multiphase systems that rely on the theory described in the equations above have been proposed. Corapcioglu and Baehr (1987) and Sleep and Sykes (1989) focused on air flow in initially unsaturated soils. In a process similar to air sparging,

the injection of steam into a saturated porous medium for remedial purposes has been modeled by Falta et al. (1992). In the next section we use the established theory described previously to explore how air flow will develop in a typical sparge installation.

Air Flow Patterns in Air Sparging Implementation

Consider a sparge injection point with a very small screen. As the air pressure is increased in the sparge well, water is displaced out through the screen. Once the well is fully dewatered, the air will be in direct contact with the screen and the porous medium just outside the screen. To initiate air movement into the water saturated medium, the steady-state pressure balance described by Equation 3 must be overcome. Examination of Equation 3 shows that the air pressure must be greater than the sum of the water pressure and the capillary pressure. Once the air pressure exceeds this sum, air enters the porous medium and migrates according to Equation 2. The value of capillary pressure at which flow begins is often called the entry pressure or bubbling pressure of the media. The entry pressure depends on grain size, packing, and other media properties (Dullien 1992).

Once the air has entered the medium, its flow behavior continues to be dictated by the pressure differences between air and water as long as the air is directly connected by continuous channels to the source of the pressure (i.e., the sparging well). The presence of this connection means that pressure in the air phase can be propagated throughout that phase. If a portion of the air stream separates from the source of pressure, it will travel through the medium as an individual bubble under the influence of the buoyancy force. In recent laboratory studies, Ji et al. (1993) found that, under steady applied pressures, the flow regime will depend on the grain size. For homogeneous coarse grained media (e.g., fine gravels and larger grain sizes), air flow occurs as discrete bubbles. However for grain sizes typical of coarse to fine sands, the flow occurs in the form of finger-like channels. These channels, once formed, remain in place as long as the air pressure is maintained. Qualitative observations indicate that increasing air pressure causes an increase in channel size as well as the formation of additional channels. The conceptual picture suggested by Ji et al. (1993) and Johnson et al. (1993) is indicated in Figure 2. Here, the injected air passes through the medium in channels of continuous air phase rather than as free bubbles as suggested by Angell (1992), Sellers and Schreiber (1992), and Marley et al. (1992). Given this type of air pathway, important considerations will be the location of the channels and their density (number of channels per unit cross section).

The problem of predicting how channels will propagate in a multifluid system has received wide study. A common problem in the oil industry is the injection of a displacing fluid, such as water or steam, to displace an existing fluid, such as oil (Falta et al. 1992). It has been widely observed in laboratory studies of this pro-

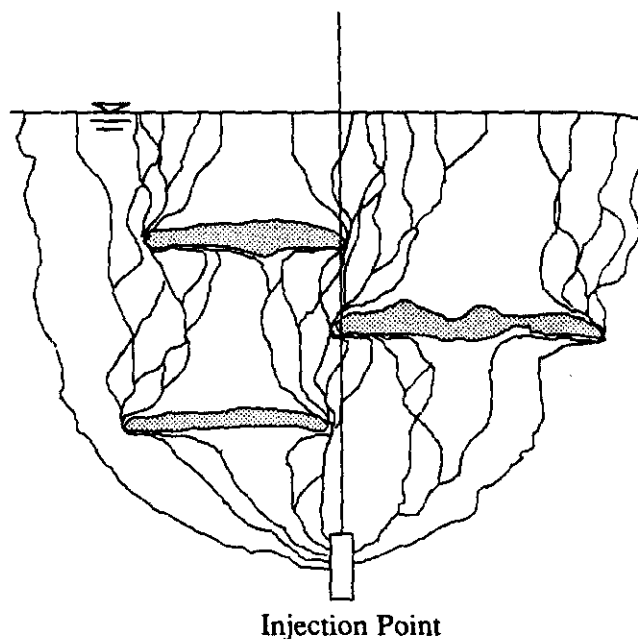


Figure 2. Schematic of air channels induced by air sparging in a media with interbedded silt lenses.

cess that a fingering phenomenon arises. As the invading front advances, preferential pathways are found through which the invading fluid flows, producing a finger-like flow pattern (Homsy 1987). It has been found that this phenomenon is most pronounced when a low viscosity fluid invades a high viscosity fluid. This adverse condition exists for the case of air sparging. Theory predicts that the way in which air channels form in a perfectly homogeneous porous medium will depend on the viscosities of each fluid and the capillary pressure.

Kueper and Frind (1988) argue that in the presence of even small heterogeneities in pore size (as would nearly always occur in natural geologic formations), the direction of finger or channel formation is dominated by the slight differences in entry pressure in adjacent media that cause the invading fluid to prefer one path over another. For a porous medium, with grain sizes such as those typically found in ground water systems, the mean radius of curvature described in Equation 4 of the interface will change depending on grain size. In an air sparging system, one can imagine a pocket of pressurized air seeking the path of least resistance for further invasion. This path will consist of the medium with the lowest entry pressure. This concept is depicted in Figure 3, where grains of porous media are surrounded by various combinations of air and water. Since at the small scale of the air pocket the water pressure around the pocket is nearly constant, the capillary pressure becomes the significant property. As the air pressure increases, the air will preferentially enter the pores with the smallest capillary pressure. Equation 4 predicts that the smallest capillary pressure will occur in pores with the largest radii of curvature. Hence, air channels will preferentially form through media with the largest pore sizes. As these media tend to have larger permeabilities, it can be expected that air flow will be correlated

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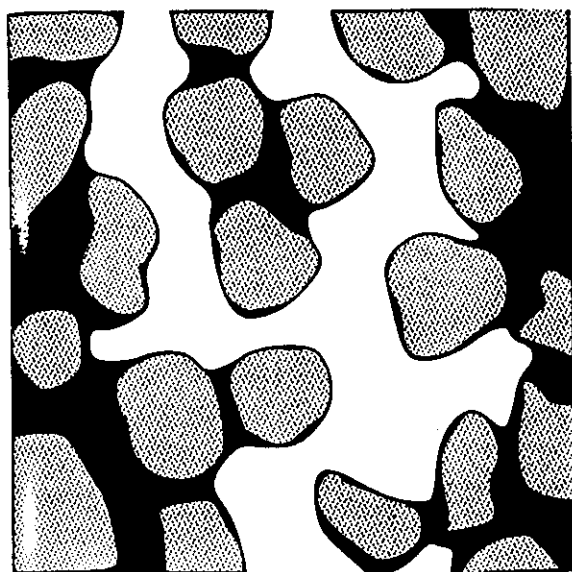


Figure 3. Schematic of channel flow at the pore scale showing interfaces between air and water.

Figure 3. Schematic of channel flow at the pore scale showing interfaces between air and water.

with pathways of high permeability and that the overall pattern of airflow will be governed by heterogeneities in permeability. The complex stratigraphy present at most sites also influences the distribution of the air channels in the porous medium. Channel density will tend to be lower for stratified media. This is due to lateral dispersion of the air when low permeability streaks are encountered (Marley et al. 1992). The tendency of air channels to respond to heterogeneities at both the pore and larger scales has been observed in the laboratory studies of Ji et al. (1993), who showed that air channels will preferentially form in coarser material.

The preceding conceptual model of air flow is central to the discussion in the remainder of this paper. The following observations on air sparging behavior assume that flow patterns of air occur in the form of discrete air channels which remain stable as long as air pressure is maintained. The location of these channels in the vicinity of the sparge point is entirely dictated by naturally occurring medium heterogeneities at the pore scale (different grain sizes) and at larger scales (different strata).

Mass Transport Mechanisms Important to Air Sparging

It has been suggested that air sparging has two significant benefits: removal of volatile organic contaminants from the subsurface and delivery of oxygen to biota for *situ* bioremediation (Johnson et al. 1993). In either case, the effectiveness of air sparging depends on mass transfer between the gaseous and aqueous phases. Determining the likely mass transfer rates of the air sparging system is complicated by the complex distribution of air channels likely to be present. For example,

consider a model for estimating equilibrium partitioning: Henry's law. This law describes equilibrium partitioning of the contaminant between the aqueous and gaseous phases at the air/water interface and can be written as:

$$C_A = C_W H_C \quad (5)$$

where C_A is the air concentration in mg/L, C_W is the water concentration in mg/L, and H_C is the dimensionless Henry's constant. This law can be used to describe the partitioning of volatile organic chemicals as well as oxygen.

In an analysis of air sparging, Wilson et al. (1992) assume that Henry's law applies to the volatile contaminants and that all water is reached by the migrating air. An examination of the validity of these assumptions yields insight into the complexity of interphase mass transfer in the context of air sparging.

Henry's law is valid when partitioning has reached equilibrium. The residence time of air, traveling in discrete channels, may be insufficient to achieve the equilibrium assumed in Equation 5 due to the high air velocities and short travel paths encountered during some sparging operations. Thus, Henry's law is a best case assumption. Perhaps more significant is consideration of whether the water concentration at the air/water interface describes the concentration in the bulk water phase.

When air channels are spaced at significant distances from each other, as might occur in the presence of interbedded strata, uniform concentration is unlikely. In this case the concentration in the water in the immediate vicinity of the air channel is reduced as the mass is volatilized from the water. To replenish the mass lost from the water at the air channel, mass transfer by diffusion and convection must occur from water not in the immediate vicinity of the air channel. Many small channels, evenly distributed, will produce smaller transport distances than a few large channels which are spaced at some distance from each other. When only a few large channels are present, less water is contacted by the injected air and the mass transfer between the water and the air may become limited by the ability of the contaminant to migrate to the air channel. Since the concentration gradient between the water and air phases at the interface is the driving force behind mass transfer, these rates will also be reduced if a high water concentration cannot be maintained at the interface.

As an example of the significance of water phase mass transport limitations, consider the theoretical diffusion calculations performed by Ji and Ahlfield (1993) under the assumption that the water concentration at the air/water interface is zero (a worst case) and that the only aqueous phase transport mechanism is molecular diffusion. These calculations show that a column of water 20 inches in radius containing a 1-inch radius air channel through its middle and contaminated with TCE at the 5000 ppb level would reach the 5 ppb level after 78 years. Increasing the channel density would decrease

column radius and decrease calculated cleanup time. The cleanup time would decrease to 1.7 years if the water column radius is reduced to 5 inches and to 27 days if the water column radius is reduced to 2 inches. While these calculations overestimate cleanup time due to the assumption that diffusion is the only aqueous phase transport mechanism, they do show that the higher the channel density (i.e., the smaller the spacing between adjacent channels), the faster the remediation. Accounting for convective transport caused by agitation would reduce the predicted remediation time, but would not change the basic conclusion that channel density is related to the rate of remediation.

Implications of the Conceptual Model for Field Application

The conceptual model generated by the preceding theoretical discussion raises questions about the practical application of air sparging. While predicting detailed behavior of air flow in heterogeneous porous media is probably impossible due to the massive data requirements, the theory of multiphase flow can be used to draw some general inferences about air flow behavior. In the following sections, we describe several implications and speculations that follow from the conceptual model of air sparging that has been described.

Definition of Radius of Influence

The term radius of influence is commonly used to describe the impact of a pumping well on saturated ground water flow. In this context, it implies that any portion of the aquifer within this radius is affected by the pumping well by a change in head and velocity gradients. This same term is commonly applied to describe the volume of water influenced by an air sparging system (Marley et al. 1992; Wilson et al. 1992; Angell et al. 1992). We believe that the definition of a radius of influence in the context of air sparging is ambiguous. The theory previously outlined predicts that, in a heterogeneous material, the density of channels will be neither uniform nor radially symmetric about a sparge injection point. Confining strata may cause the air to migrate laterally far from the sparge point (as has been reported by Marley et al. 1992), leaving untouched large volumes of water between the sparge injection point and the ultimate exit points of the air channels. Within these volumes, remediation is limited by the rate at which contaminants can migrate by diffusion or convection to the air channels.

In the context of air sparging, the radius of influence should be limited to describing an approximate indication of the average of the furthest distance traveled by the air channels from the sparge point in each radial dimension. It should not be assumed that contaminants within the radius of influence will be remediated at the same rate. The use of multiple sparge points to produce apparent overlapping influence radii does not imply that all volumes of water will be reached by the air with equal effectiveness.

Water Table Mounding and Contaminant Spreading

In many reports of field applications of air sparging, reference is made to the mounding of the water table in response to sparging (Marley et al. 1992; Brown and Fraxedas 1991; Angell et al. 1992). It has been suggested that this mounding can indicate the radius of influence of the sparging system. This may be unreliable since the mound is only formed temporarily. At the initiation of sparging, a mound will form as water pressure is increased and water is displaced. However, the mound will dissipate over time in response to the newly imposed hydraulic gradient. In the presence of air channels, water will behave in much the same fashion as in the vadose zone. It will flow in response to a gradient in hydraulic head as predicted by Equation 1 as long as the water saturation is sufficient for the relative permeability, described in Equation 2, to remain above zero. While it is conceivable that water saturation could fall below this level, this water would no longer flow into monitoring wells and its presence would not be recorded. Mounding induced by the operation of an SVE system concurrently with the sparge system has been suggested by Johnson et al. (1993) as an explanation for the mounding that is sometimes observed at sparge sites. The creation of the mound and its subsequent dissipation mean that water will be pushed away from the sparge area. Care must be taken when applying sparge systems so that sparging does not spread existing contamination.

The transient response of the water pressure to the presence of air is seen in the results of Ahlfeld et al. (1994), who sampled a number of points in the vicinity of a sparge well in a deltaic sediment formation containing medium coarse sand interbedded with numerous discontinuous lenses of low permeability silt material. In Figure 4, the change in water pressure vs. time from initiation of air sparging is depicted for two points located a radial distance of 3 feet from their vertical

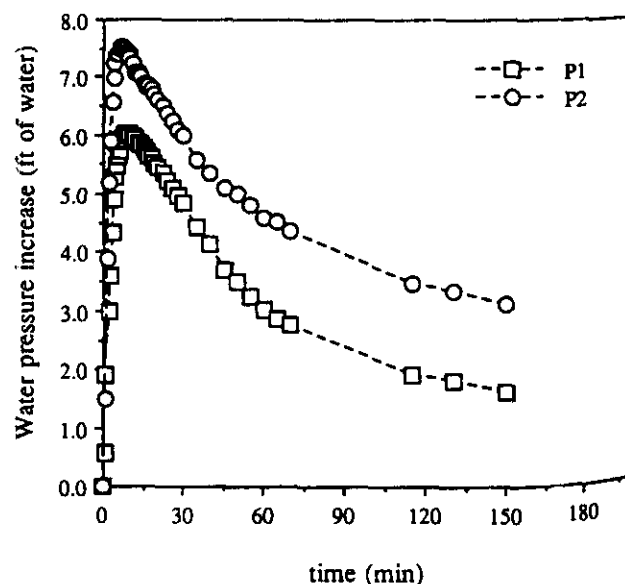


Figure 4. Change in water pressure (feet of water) at piezometer 3 feet from sparging point vs. time since initiation of sparging.

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sparge well. The P1 piezometer is at the same elevation as their 1-foot sparge screen. The P2 piezometer is about 5 feet above P1. Data points on the figure are connected with dashed line segments for graphical clarity only. The general pattern seen here is similar to that found in all measurements of water level reported by Ahlfeld et al. (1994). The water pressures increased within minutes of sparging start-up. After a short time, the water pressure began to dissipate as the water flowed away in response to the water phase pressure gradient. Water level measurements taken 24 hours after the initiation of steady air injection showed that water pressures had re-equilibrated to levels within 0.02 to 0.1 feet of those measured before sparging began. These authors also report water levels collected at the same two piezometers after the cessation of air injection. These are shown in Figure 5 where a sharp decline in water pressure is observed as the air channels collapse and water refills the resulting voids.

Sparge Wells as Interceptors

Using a line of sparge wells to intercept a migrating plume has been suggested by Wilson et al. (1992) and others. In such an arrangement, a line or cluster of sparge wells is placed downgradient of an advancing plume. The intent is that the contaminated ground water will be remediated in situ as it passes through the area in which sparging occurs. A significant complication of such a system is the likely decrease in conductivity to water that the aquifer will experience due to the presence of the air channels. Ji et al. (1993) have measured water saturations as low as 50 percent in full operation of air sparging in a laboratory setting. For many soils this would produce a reduction in conductivity of as much as 95 percent, depending on wettability. Thus, we speculate that a sparging system will produce a low permeability barrier to natural downgradient water flow. Such a low permeability barrier would, in turn, produce an increase in head upgradient of the sparge zone and a tendency for some water to flow around the sparge zone. Another concern with a sparge interception system is that some of the water that migrates through the sparging zone would not come sufficiently close to air channels for contaminant removal to occur.

If sparging does produce a low permeability barrier then care must be taken in designing a sparge interception system to avoid causing the oncoming plume to laterally disperse. Such a system could be operated in cycling mode to allow the contaminated water to enter the sparging zone and to avoid letting the plume flow around the sparging area. By staggering a grid of vertical sparge wells or placing a few rows of horizontal sparge wells or trenches, the likelihood that virtually all the contaminated water will be reached by air channels by the time it has left the sparging zone increases. Great care would have to be taken in the operation of such a system to ensure that the plume is not pushed downgradient by the operation of the system. Such a system would also have to operate as long as the plume continued to migrate in its direction.

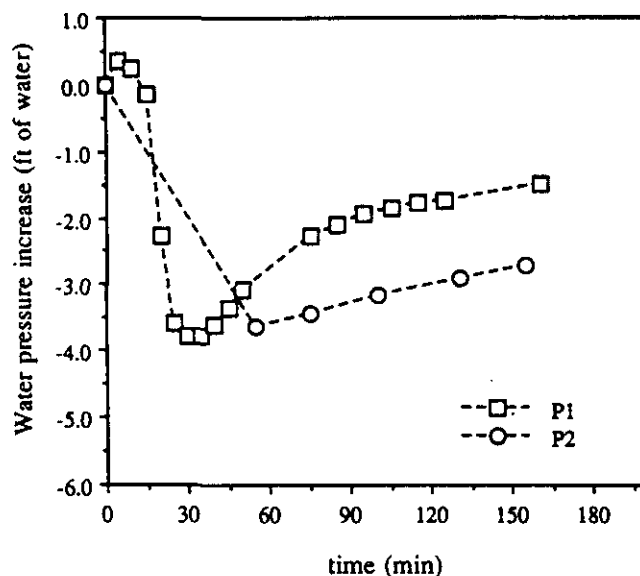


Figure 5. Change in water pressure (feet of water) at piezometer 3 feet from sparging point vs. time since cessation of sparging.

Rate of Remediation

When mass transfer is limited by the ability of the contaminant to migrate to the air channel, the location, number, and density of air channels will influence the ability of the sparging system to volatilize contaminants. At first, the rate of contaminant recovery may be rapid as water in the immediate vicinity of air channels is stripped of contaminants; however, we speculate it is likely to slow as the interphase transfer of contaminants becomes limited by aqueous phase transport. The presence of a nonaqueous phase liquid (NAPL) contaminant can seriously impede the efficiency of air sparging. If no air channel exists near a NAPL pool or ganglia, then the pathway for remediation is complicated. First the contaminant would have to dissolve into the aqueous phase, then it would have to migrate to the air channel via convection and diffusion. These are some of the same limitations on the rate of remediation of NAPLs that have plagued pump-and-treat techniques. This sort of behavior has been observed at several sites where a tailing off of effluent concentrations is reported (Marley et al. 1992; Angell et al. 1992; Hennes and Feenstra 1993).

The presence of diffusion limitations may severely affect the efficiency of air sparging. In comparison to conventional pump-and-treat remediation, Angell (1992) has identified two benefits of air sparging: increased mass transfer and volumetric flow of air. When contaminant removal (or oxygen delivery) is limited by the ability of the contaminant to migrate through the water phase to the air phase, these benefits are lost. Two operational techniques, cycling and special screen configurations, have been suggested to address this problem.

Cycling

Several reported applications of air sparging have operated in a cycling fashion with the air injection varied

over a range of pressures or simply turned on and off with some frequency (Marley et al. 1992; Johnson et al. 1993). The conceptual model indicates that channels will form in pathways of largest pore diameters. As long as the pore structure of the porous medium remains the same from one cycle to the next (which may not be the case if significant medium fracturing occurs due to very high air pressures), we believe that the air pathways formed during each successive cycle of operation should be nearly the same. Cycling should not be expected to cause the water to migrate in new directions, especially if the pathway formation is dominated by large scale heterogeneities. We speculate that the primary benefit of cycling will likely be to induce agitation and mixing of the water as the air channels form and collapse during each cycle. This should help to alleviate the limitation of mass transport through the bulk water phase. In the case of an interception system, cycling should allow ground water flow to resume its natural rate and direction and bring new contaminated water into the area influenced by the sparge operation.

Screen Design

Screens or injection points with special features, such as diffusers, have been suggested by Marley et al. (1992) and Sellers and Schreiber (1992). The intent of such devices is to force an initial geometric configuration of the air stream which will influence its subsequent migration through the formation. While such a device may affect air behavior within several inches of the screen, we speculate that as soon as the air stream gets a short distance into the formation the structure of the natural media will dominate. If the air stream induced by the screen has a diameter larger than the pore openings, then the air will coalesce and its subsequent migration will be dominated by in situ media structure. Based on this argument, short slotted screens used for water supply or monitoring wells are adequate for introducing air into the saturated zone.

Summary and Conclusions

In the heterogeneous, stratified formations in which sparging is often applied, the pattern of air movement through the subsurface is complex. This complexity is largely driven by variations in grain size, capillary resistance, and intrinsic permeability of the porous media. The presence of air in the form of discrete channels affects the flow of water and subsequent migration of contaminants.

A number of implications follow from an examination of the well-developed theory of multiphase flow in porous media. The complicated hydrodynamics of air and water flow mean that the common understanding of a radius of influence about a sparge point must be carefully applied. The presence of air channels impedes but does not stop the flow of water. The mounding of water around a sparge point is a transient phenomenon which should decay over time. The natural ground water flow through a sparged zone of an aquifer will be slowed and diverted by the air channels.

In the presence of air channels, the rate of mass transfer will be limited by either kinetics of the mass transfer at the interface or by the rate of transport of the contaminant through the bulk water phase to the air/water interface. The limiting transport mechanism depends on the type of contaminant, density of air channels, and site-specific permeability characteristics.

Several techniques may prove useful for alleviating these problems, including operational cycling and placement of air injection screens in the vicinity of identified heterogeneous strata. However, in some circumstances air sparging will not be an effective method for efficient remediation. More research and field experience are needed to better quantify the range of conditions under which air sparging will be successful.

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Biographical Sketches

David P. Ahlfeld is an associate professor in the Department of Civil Engineering at the University of Connecticut (Box U-37, Dept. of Civil Engineering, The University of Connecticut, Storrs, CT 06269). He holds a Ph.D. in civil engineering from Princeton University. His research interests include ground water modeling and remediation.

Amine Dahmani is engineering project manager in the Environmental Research Institute at the University of Connecticut (Box U-210, Environmental Research Institute, The University of Connecticut, Storrs, CT 06269). He holds a Ph.D. in petroleum engineering from Louisiana State University. His primary research activities are in soil contamination characterization, leaching potential of pollutants, and innovative ground water remediation technologies.

Wei Ji is a Ph.D. student in the Department of Civil Engineering at the University of Connecticut (Box U-37, Dept. of Civil Engineering, The University of Connecticut, Storrs, CT 06269). He has a B.S. in fluid mechanics and an M.S. in computational mathematics from Peking University, China. His research interests include fundamental studies of conceptual models, modeling of multiphase fluid flow, and ground water contaminant transport.

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Distribution of Redox-Sensitive Groundwater Quality Parameters Downgradient of a Landfill (Grindsted, Denmark)

POUL L. BJERG,* KIRSTEN RÜGGE,
JØRN K. PEDERSEN, AND
THOMAS H. CHRISTENSEN

*Institute of Environmental Science and Engineering,
Groundwater Research Centre, Technical University of
Denmark, Building 115, DK-2800, Denmark*

The leachate plume stretching 300 m downgradient from the Grindsted Landfill (Denmark) has been characterized in terms of redox-sensitive groundwater quality parameters along two longitudinal transects (285 samples). Variations in the levels of methane, sulfide, iron(II), manganese(II), ammonium, dinitrogen oxide, nitrite, nitrate, and oxygen in the groundwater samples indicate that methane production, sulfate reduction, iron reduction, manganese reduction, and nitrate reduction take place in the plume. Adjacent to the landfill, methanogenic and sulfate-reducing zones were identified, while aerobic environments were identified furthest away from the landfill. In between, different redox environments, including apparent transition zones, were identified in a sequence in accordance with the thermodynamic principles. The redox zones are believed to constitute an important chemical framework for the attenuation processes in the plume.

Introduction

Old landfills without leachate collection systems are numerous all over the world. Leakage of inorganic and organic pollutants from these landfills may influence the groundwater quality and thereby be a risk to drinking water resources. In order to evaluate this risk, to design groundwater detection monitoring programs, and to perform remedial action, a detailed understanding of the attenuation processes in leachate plumes is highly needed.

The entrance of strongly reduced landfill leachate into an aquifer may lead to the development of different redox environments in the plume depending, among many factors, on the redox capacities and reactivities of the reduced and oxidized compounds in the leachate and in the aquifer. The attenuation processes in the plume will most likely, for many pollutants, depend on the redox environments in the plume and in some cases also contribute to the development of these redox environments. In an actual leachate plume, the redox environments are supposed to develop over many years as a result of interactions between, for example, redox processes, dissolution-precipitation processes, ion exchange processes, and dilution. Redox environments in leachate plumes, as a chemical framework for the attenuation processes, have only gained little attention in the literature (see the review in ref 1). Some reports on leachate plumes provide information indicating that different redox environments were present in the plumes (2, 3), but only one report seems to exist (4) specifically addressing the governing redox environments.

The purpose of this study was to identify the distribution of redox-sensitive groundwater constituents in an actual landfill leachate plume and to identify the governing redox environments in the plume. The fate of organic chemicals originating from municipal waste (5) and from pharmaceutical waste (6) is discussed in other papers with reference here to the described redox environments.

Materials and Methods

Grindsted Landfill. Grindsted Landfill is located on the ground surface in a flat landscape with heath and small woods (Figure 1a). The maximum height of the landfill is 8 m, and the average height is 5 m. The landfill received waste in the period 1930-1977, but the main part of the waste was landfilled in the 1960s and 1970s (7). The landfill covers approximately 10 ha and contains about 500 000 m³ of waste. Neither liners nor leachate collection system exist. Parts of the landfill have received primarily household waste, while other parts have received mainly demolition waste. A certain part of the landfill has been used for industrial waste, disposed of in lagoons. However, small amounts of chemical waste may have been disposed of all over the landfill area.

Hydrogeology. The landfill is located on a glacial outwash plain. The investigated aquifer has a depth of approximately 10-12 m and consists of two sandy layers: 5-7 m of glaciofluvial sand (quaternary period) overlying a micaceous sand (miocene period) with a thickness of approximately 4-6 m. The glaciofluvial sediments of the

* Corresponding author.

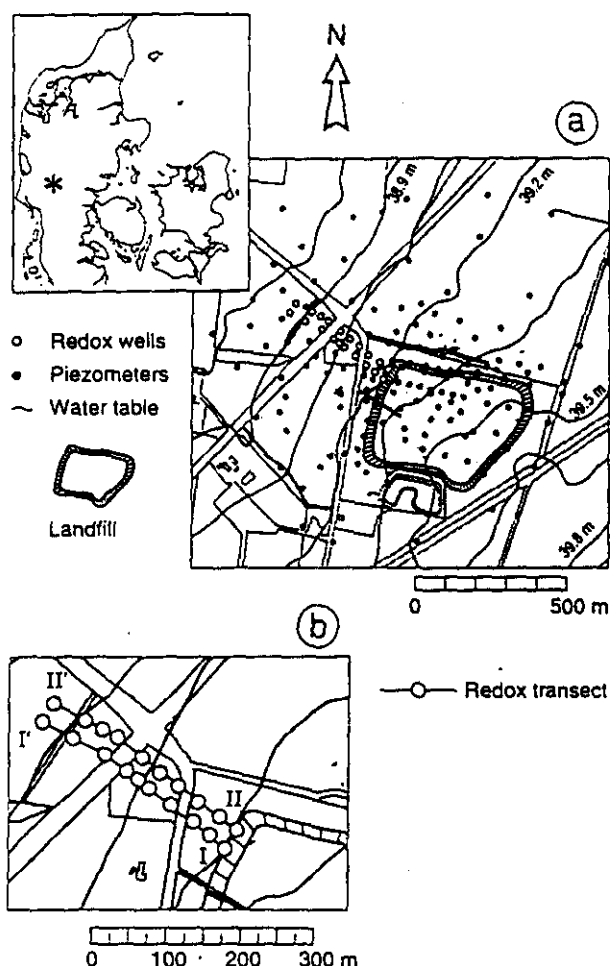


FIGURE 1. Map of the Grindsted Landfill area, contour map of isopotential curves (December 2, 1992), and location of wells in transects I and II.

aquifer are composed of medium and coarse grained sand and gravel. The micaceous aquifer material is well sorted and consists of fine or medium grained sand. The two sand deposits are in some areas separated by a thin strata of silt or clay identified in sediment cores. The aquifer is downward limited by a miocene silt/clay layer.

The annual rainwater infiltration is approximately 400 mm yr⁻¹. The overall groundwater flow direction has been determined by water level measurements in 165 piezometers installed in the upper aquifer (Figure 1). The water table is located 1–3 m below ground surface. In the autumn of 1992, the water table fluctuation was approximately 1 m, but the variation was uniform over the study area. A water table contour map is shown in Figure 1. The isopotential curves are evenly spaced, indicating fairly homogeneous hydraulic properties of the aquifer. The overall groundwater flow direction is westerly, but the isopotential lines are semicircular, indicating a diverging flow. The average hydraulic gradient is 0.0012.

The hydraulic properties of the aquifer were investigated by mini-slug tests (according to ref 8). The geometric mean of the hydraulic conductivity of the glaciofluvial sand is 4.6×10^{-4} m s⁻¹, and the miocene sand has a geometric mean of 0.9×10^{-4} m s⁻¹. The variance of the log-normalized hydraulic conductivities is 0.47 for the upper layer and 0.41 for the lower layer. The small variance of the hydraulic conductivity of both layers indicates a homogenous aquifer. The data on hydraulic gradient and hydraulic conductivity

yield, assuming an effective porosity of 0.33, an average linear groundwater pore flow velocity of 50 m yr⁻¹ for the glaciofluvial sand and 10 m yr⁻¹ for the miocene sand.

Wells. The groundwater quality was mapped in terms of two transects in the flow direction from the landfill (Figure 1b). The horizontal distance between the two longitudinal transects was 30 m. In each transect (300 m), 10 driven wells were established with a horizontal distance of approximately 30 m. Groundwater samples were collected over depth with 0.5-m increments. The wells consisted of iron pipes (diameter of 2.0 cm), with a 10-cm screen, driven into the ground by a pneumatic hammer. A Teflon check valve was placed over the screen. In transect I, seven additional permanent wells, each with three or four sampling points over depth, were established decreasing the horizontal spacing between the wells to 8 m near the landfill.

Groundwater Sampling and Analysis. Groundwater sampling was performed by nitrogen pressure through a Teflon tube lowered into the well. The Teflon tube was connected directly to a multiboard device, where pH, oxygen, temperature, redox potential, and chemical conductivity were measured by electrodes in a flow cell. For details on the sampling system and instrumentation, see Lyngkilde and Christensen (4). In total, 285 groundwater samples were obtained.

All samples, except for methane, sulfide, and dinitrogen oxide analysis, were pressure-filtered by nitrogen through a 0.1- μ m membrane filter. Samples for methane and sulfide were, to avoid volatilization, obtained directly by a syringe. Samples were preserved immediately after sampling.

The water samples were analyzed in the laboratory for chloride, nitrate, nitrite, sulfate, ammonium, methane, dinitrogen oxide, dissolved organic matter expressed as nonvolatile organic carbon (NVOC), dissolved iron, and manganese. Preservation, sampling containers, and analytical methods were similar to the methods applied by ref 4, except for sulfate, sulfide, and dinitrogen oxide as described below.

Sulfate was determined by a turbidimetric method (9). Samples for sulfide analysis were collected by a syringe and mixed with a buffer solution (pH 11, constant ionic strength) to obtain constant pH. Sulfide was quantified by a selective sulfide electrode (Radiometer F1212S). A Calomel electrode (Radiometer K711) was used as reference electrode in a dark cell to avoid photographic interference. Samples (6.5 mL) for analysis of dinitrogen oxide (N₂O) were transferred from the syringe to an evacuated Venoject blood sample vial (13 mL) and preserved with three drops of concentrated sulfuric acid. The quantification was done by injecting a gas sample (0.5 mL) from the headspace on a Carlo Erba gas chromatograph, Vega Series 2, equipped with an electron capture detector and a packed column, Porapak Q. The response was quantified based on a standard curve from the responses of external standards made by injecting known volumes of N₂O into vials containing distilled water and concentrated sulfuric acid.

Leachate Quality. The leachate entering the transects originates from an area of the landfill where mainly municipal waste and some liquid waste from a pharmaceutical industry have been landfilled. The leachate measured in the top of the groundwater just below the landfilled waste has been described as dilute methanogenic phase leachate (7) with NVOC in the order of 20–200 mg L⁻¹ and a BOD₅/COD ratio of 0.03–0.20 (BOD₅, biological

TABLE 1

Background Groundwater Quality and Composition of Leachate/Groundwater Mixture at Western Border (Northern Part) of the Landfill Where Transects Begin

parameter	unit	background groundwater quality	leachate/groundwater quality
methane	mg of CH ₄ L ⁻¹	0	7-43
sulfide	mg of S ²⁻ L ⁻¹	<0.1	0.01-2.3
sulfate	mg of SO ₄ ²⁻ -S L ⁻¹	3-14	<1*
iron(II)	mg of Fe ²⁺ L ⁻¹	0-0.3	32-142*
manganese(II)	mg of Mn ²⁺ L ⁻¹	0-0.1	2-5.5
ammonium	mg of NH ₄ ⁺ -N L ⁻¹	<0.1	30-116
nitrite	mg of NO ₂ ⁻ -N L ⁻¹	<0.05	<0.05
nitrate	mg of NO ₃ ⁻ -N L ⁻¹	0.5-5	<0.05
oxygen	mg of O ₂ L ⁻¹	1-8	0
NVOC	mg of C L ⁻¹	1-3	80-120
chloride	mg of Cl ⁻ L ⁻¹	12-30	29-143
pH		4.5-6	6.5-7

* Excluding a few sampling points where extremely high sulfate concentrations have been observed (see text).

oxygen demand determined after 5 days; COD, chemical oxygen demand). No records exist on the development in leachate quality over time.

The composition of the leachate/groundwater mixture at the border of the landfill at the beginning of the transects has been studied in detail and revealed a fairly homogeneous distribution. The composition of the leachate/groundwater mixture in this area is summarized in Table 1.

Background Groundwater Quality. The background groundwater quality determined in nearby wells outside the pollution plume is shown in Table 1. The unpolluted aquifer is aerobic (1-7 mg of O₂ L⁻¹) with moderate concentrations of nitrate compared to nitrate concentration in farmland areas nearby. The pH is low (4.5-6) in the noncalcareous aquifer and shows no dependency over depth. Chloride concentrations are around 15-20 mg L⁻¹. The content of NVOC ranges from 1 to 3 mg of C L⁻¹. Reduced species as iron(II), manganese(II), ammonium, and sulfide are not present in significant concentrations.

Results and Discussion

The two longitudinal vertical transects were placed along flow lines based on the described flow field. As seen in Figure 1, the transects are almost perpendicular to the isopotential curves. The measured distribution of compounds along transect I are presented in Figure 2, and a selection of compounds along transect II is shown in Figure 3. The transects show a similar pattern for most compounds. The presentation of the results primarily refers to transect I, but where important differences are found, transect II is included in the discussion as well.

Characteristics of the Leachate Plume. The leachate plume contains elevated concentrations of chloride, NVOC, methane, ammonium, iron(II), and manganese(II). The plume is recognizable for about 200-250 m. The shallow aquifer is polluted over the entire depth at the border of the landfill. Presumably, the hydrology and hydrogeology of the landfill causes a mixing of leachate and groundwater beneath and at the border of the landfill (7). Gradually a zone with uncontaminated infiltration water develops above the plume. The boundary between the plume and the ambient water is very sharp, and apparently the vertical

dispersion is rather limited. The distribution of the compounds seems to be partly influenced by the intermediate silt/clay strata (34-34.5 m above sea level) in the upper part of the aquifer, which where present seems to limit the vertical mixing in the aquifer. The distinct distribution of the contaminants over depth is in accordance with other landfill leachate plume studies (e.g., refs 4, 10, and 11).

The pH values in the leachate are near neutral (pH 6.5-7) and strongly buffered with high alkalinity values. pH and alkalinity (not shown) decrease along the two transects and, in a distance of 150 m, approach the background level.

The chloride concentrations in the plume are low compared to other studies. Within the first 60-70 m of the plume, chloride is only very slightly diluted. Between 90 and 120 m, the concentrations decrease rapidly in the lower part of the aquifer, and at 120 m the contrast between background levels and leachate-affected groundwater is minor with respect to chloride. This is difficult to explain since ammonium and oxygen still clearly indicate that the aquifer is affected by the leachate plume as far as 200-250 m from the landfill. Variations over time in the composition of the leachate at the source may be an explanation. In the upper part of the aquifer (3-5 m below ground surface) at a distance of 90 m from the landfill, high chloride concentrations were measured in a few sampling points. Farther downgradient of the landfill, high chloride concentrations, even higher than in the vicinity of the landfill, have been observed in all depths. This could be due to road salt contamination. Information provided by the local road authorities supports that road salt, used for deicing during the winter, may be the most probable explanation. Road salt has been applied since 1990 at the road crossing 70 m from the landfill (Figure 1). The highway crossing the transects in a distance of 180 m has been heavily salted for more than 10 yr. Chloride has often been used in other studies as a dilution indicator parameter, assuming that chloride acts as a conservative tracer (3, 12, 13). In this case, the distribution of chloride seems to be significantly affected by the road salting, and use of chloride as a tracer may be dubious at distances larger than 60-70 m in the upper part of the aquifer. In the lower part of the aquifer, chloride may be applicable as a tracer up to 120 m from the landfill.

The NVOC concentrations in the plume decreases with distance from the landfill. At 170 m from the landfill, the concentrations correspond to the background concentration of 1-3 mg of C L⁻¹. Attenuation of NVOC may be governed by dilution, sorption, and degradation. Dilution (based on chloride) seems insignificant within the first 60-70 m but may reduce the NVOC concentration at further distances from the landfill. Sorption of leachate NVOC onto aquifer sediment is not very significant according to the literature (1). The rapid decrease of NVOC may then be explained partly by dilution and partly by degradation, which will be further discussed in ref 5.

Distribution of Redox-Sensitive Constituents. Methane and carbon dioxide are produced in a methanogenic phase landfill. The ratio between CH₄ and CO₂ is usually around 1-2.3 corresponding to a partial pressure of CH₄ of 0.5-0.7 (14). The solubility of methane is 18.8 mg L⁻¹ in water (8 °C, 1 atm, and a partial pressure of 0.6). In the vicinity of the landfill, we observed methane concentrations in a range from 7 to 43 mg L⁻¹. The very high methane content in a few samples could be an artifact caused by bubbles formed

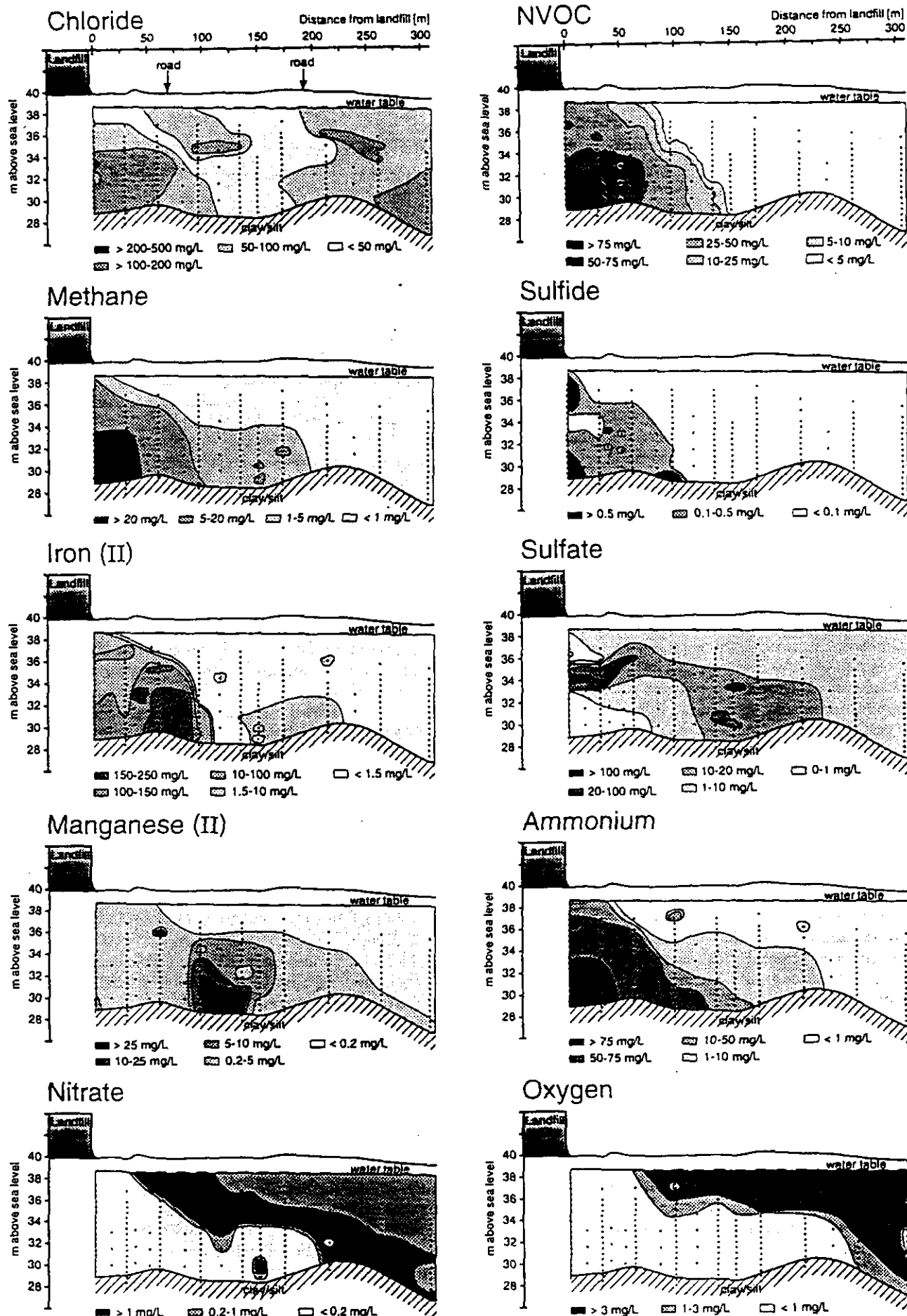


FIGURE 2. Distribution of chloride, NVOC, methane, sulfide, sulfate, iron(II), manganese(II), ammonium, nitrate, and oxygen in groundwater downgradient of Grindstedt Landfill transect I. NVOC is in mg of C L⁻¹. Nitrate and ammonium are in mg of N L⁻¹. Sulfate is in mg of S L⁻¹.

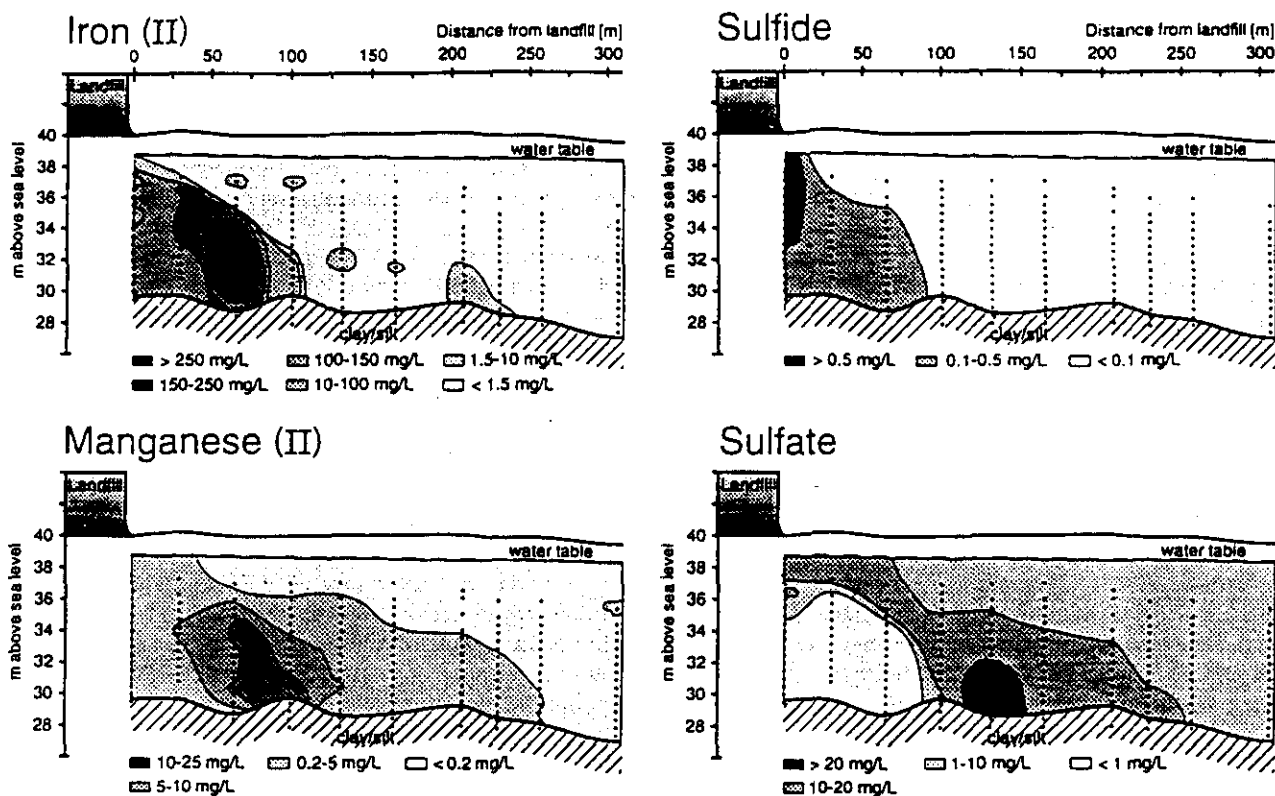


FIGURE 3. Distribution of sulfide, sulfate, iron(II), and manganese(II) downgradient of Grindsted Landfill transect II. Sulfate is in mg of S L⁻¹.

during sampling. The average methane concentrations are higher than the solubility at the groundwater table (or inside the landfill) and indicates methane production in the aquifer, where the pressure is higher than at the surface allowing for a higher solubility. We do not see any increase in methane concentrations with distance, probably because the solubility of methane then would be exceeded. However, it cannot be excluded that the measured methane in the plume partly was produced in the aquifer beneath the landfill. The very high methane concentrations persist for more than 50 m, and methane is detectable in concentrations of more than 1 mg L⁻¹ throughout the anaerobic part of the plume.

Sulfide is detected close to the landfill (0–60 m) in low concentrations (<2 mg L⁻¹). The sulfide plume at the landfill border can with respect to depth be divided in transect I into two distinct zones separated by a zone without sulfide present. The distribution of **sulfate** is complementary to the distribution of sulfide. Where high sulfide concentrations were detected, we found low sulfate concentrations and vice versa. The zone with high concentrations of sulfate may originate from a certain part of the landfill where waste rich in sulfate has been disposed (e.g., demolition waste). Unfortunately, this complex leaching pattern complicates the interpretation of the sulfate and sulfide distribution in terms of mass balances. Another complicating factor is the possibility of sulfide precipitates (see later). In transect II, a more homogeneous distribution is observed (Figure 3) characterized by low sulfate concentrations and the presence of sulfide close to the landfill. Sulfate reduction is indicated by the presence of sulfide and low concentrations of sulfate in the same areas. The general increase of sulfate about 90 m from the landfill could be a result of oxidation processes (see the section on manganese), but the uneven leaching of sulfate into the plume cannot be ruled out as an explanation.

Methane and sulfide have been identified in the same areas. In the literature, it has been shown that methane production and sulfate reduction in some cases exclude each other, but also the coexistence of methanogens and sulfate reducers has been observed (15, 16). Probably the coexistence of methane and sulfide could also be due to mixing of water during the sampling of the well from microenvironments with different redox status.

Dissolved **iron** is present in high concentrations at the border of the landfill. Dissolved iron is practically synonymous with ferrous iron. The concentrations increase 30–60 m from the landfill, which could be explained by variations in the source or by reduction of solid Fe(III) minerals on the sediment to aqueous iron(II). Iron reduction could be an active process now or in the past, where iron(II) may have been generated closer to the landfill and then migrated to the current position. In transect II, the increase of dissolved iron is very significant (Figure 3). Here concentrations of more than 400 mg of Fe L⁻¹ were measured 60 m from the landfill border. Variations in the source over time are most likely not the case considering that no other leachate pollutants, e.g., chloride and ammonium, show such large increases in the plume. Presumably the explanation is iron reduction. The iron reduction process is considered mainly to be a microbially mediated process, but chemical reduction by sulfides also has been documented [see reviews by Lovley (17, 18)]. The latter could be an important sink for sulfide as plume in column studies (19), but in a full-scale pollution plume the relative importance of the two processes is hard to evaluate. Over the distance (30–60 m from the landfill) where iron(II) increases significantly, NVOC decreases about 25 mg L⁻¹ (2 mmol L⁻¹). Using the stoichiometry for iron reduction by organic matter ($\text{CH}_2\text{O} + 4\text{Fe}(\text{OH})_3 + 8\text{H}^+ \rightarrow 4\text{Fe}^{2+} + 11\text{H}_2\text{O} + \text{CO}_2$) and assuming that the decrease in NVOC solely was caused by the reduction of iron(III), the decrease

of NVOC will create 8 mmol L^{-1} (450 mg L^{-1}) dissolved iron compared to an observed increase of around $100\text{--}300 \text{ mg L}^{-1}$. This seems reasonable because the created iron(II) probably will be partly removed from the groundwater by ion exchange or precipitation. Thus iron reduction is a likely process in the plume as also previously shown at the Vejen Landfill (4) and indicated at a number of landfill studies (see review in ref 1). Because ferrous iron is strongly active in ion exchange, it is difficult to conclude to what extent the current distribution of ferrous iron is due to past or to on-going iron-reduction activity. Further away from the landfill, the dissolved iron concentrations diminish rapidly, supposedly due to precipitation (see discussion later), ion exchange with the aquifer material, and dilution.

Both sulfide and methane were detected in areas with high concentrations of iron(II). Investigations of river sediments (20) and aquifer material (21) strongly indicate that iron reduction inhibits sulfate reduction and methane production. Therefore, the simultaneous presence of methane, sulfide, and iron in this part of the plume may be a result of a combination of methane and sulfide migration and production of ferrous iron or a combination of methane production and sulfate reduction and ferrous iron generated earlier at higher redox levels and retarded in the aquifer.

Both sulfide and methane were detected in areas that were highly influenced by iron reduction. Investigations of river sediments (20) and aquifer material (21) strongly indicate that iron reduction inhibits sulfate reduction and methane production. Therefore, it seems likely that the presence of sulfide and methane in this part of the plume is a result of migration and not due to methane production or sulfate reduction.

The distribution of dissolved manganese in the plume is characterized by elevated concentrations in an area $90\text{--}140 \text{ m}$ from the landfill. In transect II, a similar pattern can be observed, but the concentrations are lower and the maximum values are found slightly closer to the landfill. Compared to other investigations (2–4), the concentrations of dissolved manganese are extremely high (up to 40 mg L^{-1}). The significant manganese concentrations could be due to microbial reduction of manganese oxides by oxidation of organic matter or by chemical oxidation of iron(II) or sulfide(II) (18, 22). Iron and manganese have been found in the same areas of the plume in transect II, which supports that chemical reduction by iron(II) may take place. However, the end product in both cases is dissolved manganese(II) (21). In the literature, especially manganese oxides are reported to be reactive with reduced sulfur compounds (18), but the importance of this process is difficult to evaluate. In a column study (19), it has been shown that the relative importance of the process compared to microbial iron reduction was minor because only the most reactive part of the manganese oxides was active in the reaction. In our study, the oxidation of sulfur compounds is indicated by the increasing sulfate concentrations just downgradient of the area with elevated manganese concentrations. Calculations based on the increase of dissolved manganese(II) indicate that the observed increase of sulfate is larger than what can be accounted for by manganese(IV) reduction. However, the formed manganese may be underestimated because the increase in the aqueous phase probably will also cause an increase in the solid phase, which is not accounted for in this context. Another contribution to the increasing sulfate concentra-

tions may be oxidation of pharmaceutical sulfur compounds disposed of at the landfill (6) or variations in the source.

The previous sections have shown the possible importance of solid phases in the anaerobic zones of the plume, e.g., the presence of reduced iron(II) and manganese(II) in high concentrations may cause precipitation of sulfides resulting in very low concentrations of sulfide. In order to evaluate this aspect, calculations by means of the geochemical speciation model Minteqa2 (23) have been performed. The original thermodynamic data base has been changed and extended with the stability constants for iron and manganese carbonates proposed by ref 24. These constants may be crucial for anaerobic water rich in iron(II) and carbonate for the geochemical speciation (19, 24). The calculations generally indicate supersaturation with respect to FeS (mackinawite/monosulfide) at the border of the landfill to a distance of around $60\text{--}90 \text{ m}$ from the landfill, where the concentrations of iron(II) and sulfide fate rapidly. Likewise, the solubility products for FeCO_3 (siderite), CaCO_3 (calcium carbonate) and $\text{CaMg}(\text{CO}_3)_2$ (dolomite), and MnCO_3 (rhodocrosite) are exceeded in this part of the plume. In the case of rhodocrosite, supersaturation is also found in the zone high in manganese ($90\text{--}120 \text{ m}$ from the landfill). Supersaturation has been reported previously in landfill pollution plumes studies (2, 3) and indicates slow kinetics of the precipitation processes. The existence of organic iron complexes (increasing the solubility), which are not taken into account in the model calculations, however may also contribute to the apparent supersaturation.

The speciation showed for major cations the presence of inorganic complexes, especially for calcium and magnesium, but in all cases the free ion dominated (normally $80\text{--}90\%$ of the total concentration of the compound). For iron and manganese, hydrogen carbonate complexes were significant. Here approximately 50% of the iron and $10\text{--}40\%$ of the manganese were complexed as hydrogen carbonates.

The ammonium plume has a pattern similar to the methane plume, but with a slightly larger extent. The 1 mg L^{-1} isocontour line closely follows the 1 mg L^{-1} oxygen limit. This indicates nitrification in the interface between the anaerobic and aerobic parts of the plume. However, lower ammonium concentrations closer to the landfill indicate that cation exchange processes may attenuate ammonium in the anaerobic part of the plume.

Nitrate is absent close to the landfill. Gradually, the extent of nitrate increases, and about 200 m from the landfill, nitrate is present in the entire profile. However, the fact that nitrate (without any oxygen present) envelopes the plume indicates that nitrate reduction may take place in the interface. This was further investigated by measuring dinitrogen oxide in the groundwater samples, which is an intermediate compound in the denitrification process (25, 26). Dinitrogen oxide (not shown) was detected in elevated concentrations in an area from around 90 m downgradient the landfill. The highest concentrations ($80\text{--}120 \mu\text{g}$ of $\text{N}_2\text{O-N L}^{-1}$) were found around $150\text{--}200 \text{ m}$ from the landfill border at $6\text{--}10 \text{ m}$ depth below surface. This level is consistent with values given by Ronen et al. (27) for a sewage-polluted aquifer. Nitrite was observed in very low concentrations in a few samples. The carbon source for denitrification may be dissolved organic matter from the landfill, solid organic matter associated with the aquifer material, or methane. The organic matter from the landfill

TABLE 2

Criteria Used for Assignment of Redox Status to Groundwater Samples (Grindsted Landfill, Denmark)^a

parameter	methanogenic	sulfate reducing	iron reducing	manganese reducing	nitrate reducing	aerobic
methane	>25					<1
sulfide		>0.1			<0.1	<0.1
sulfate						
iron(II)	<150	<150	>150	<10	<10	<1.5
manganese(II)	<5	<5	<5	>5	<0.2	<0.2
ammonium						<1
dinitrogen oxide	<1	<1	<1	<1	>1	
nitrite	<0.1	<0.1	<0.1	<0.1	>0.1	<0.1
nitrate	<0.2	<0.2	<0.2	<0.2		
oxygen	<1	<1	<1	<1	<1	>1

^a All units are in mg L⁻¹, except dinitrogen oxide which is in $\mu\text{g L}^{-1}$. Nitrate, nitrite, and ammonium are in mg of N L⁻¹. Sulfate is in mg of S L⁻¹.

has at this distance diminished to background concentrations, while methane still is present. This may indicate the concept that methane could be the carbon source for nitrate reduction, and actually methane disappears in this area.

The nitrate distribution indicates that nitrate is formed at the outer edge of the ammonium plume as discussed above. When methane is exhausted, denitrification is probably limited, and the concentration of nitrate formed from oxidation of ammonium may increase. Diffusion of oxygen into this area may allow for nitrification of ammonium.

The unpolluted aquifer is aerobic, but due to the strongly anaerobic leachate, oxygen is depleted in the plume. Oxygen penetrates gradually deeper and deeper, and the bottom part of the aquifer is again oxidized at 250–300 m downgradient of the landfill. There are a few sampling points 300 m from the landfill border where oxygen is absent. This is probably due to a small bog where zones with low hydraulic conductivity and high contents of solid organic matter exist. In transect II, oxygen is present in all sampling points over depth at this distance.

Redox Zonation. The distribution of the redox-sensitive groundwater parameters indicates that several redox processes take place in the plume; some areas may be dominated by one redox process while other areas may host several concurrent redox processes. Lyngkilde and Christensen (4) developed for the leachate plume at Vejen Landfill a redox criteria scheme for assignment of redox status to groundwater samples. The redox criteria scheme was developed according to thermodynamic principles and local conditions in terms of leachate and groundwater composition. In this study, we tried to apply the same redox criteria to the Grindsted Landfill leachate plume because we see the approach as a convenient way to identify the governing redox conditions in a full-scale pollution plume. The redox criteria were completely transferable for the aerobic part and partly transferable for the denitrifying part of the aquifer. In the strongly anaerobic part of the aquifer, the criteria noted by ref 4 were violated due to very high concentrations of methane, iron, and manganese and the presence of sulfide. This points out that a redox criteria scheme for the assignment of redox status based on concentrations of redox-sensitive groundwater parameters in a dynamic system is site specific and shall pay attention to the actual conditions at the field site, especially in anaerobic redox environments where sediment bound oxidants and precipitation products play a significant role. For engineering practice, evaluation of redox conditions in the field based on measurements of only a few

Redox zones:

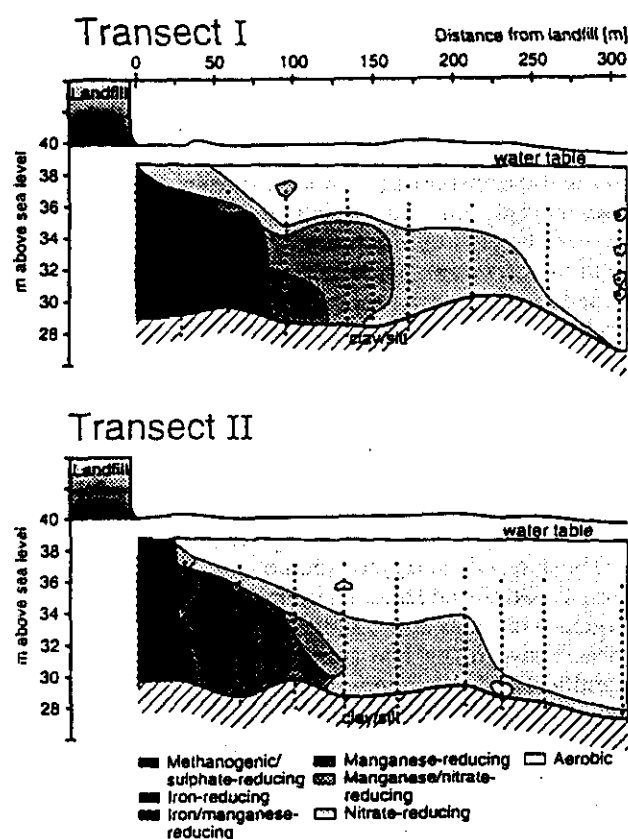


FIGURE 4. Proposed distribution of redox zones downgradient of Grindsted Landfill transects I and II.

redox-sensitive groundwater parameters in a few wells should only be taken as indicative, especially under anaerobic conditions.

A redox criteria scheme for the Grindsted Landfill leachate plume is proposed in Table 2. It should be emphasized that many of the redox-sensitive compounds involved in the criteria after formation may be transported or take part in dissolution/precipitation processes or cation exchange as described previously. The criteria proposed in Table 2 were applied to the groundwater samples from the two transects. More than 90% of the samples complied with the criteria. The remaining samples were individually evaluated, and a redox status was assigned paying attention to the redox status of the surrounding samples. In Figure 4, interpreted redox zones for transects I and II are proposed based on the criteria values in Table 2.

The proposed redox zone patterns of transects I and II are in general very similar. The redox sequence is thermodynamically sound with methanogenic/sulfate-reducing conditions close to the landfill followed by iron-reducing, manganese-reducing, denitrifying, and finally aerobic conditions 250 m from the landfill. The different zones show some overlapping in the reduced part of the plume. It has not been possible to distinguish between the methanogenic zone and the sulfate-reducing zone. Likewise, the iron-reducing zone and the manganese-reducing zone to a large extent overlap each other in transect II. In transect I, the areas with manganese reduction and denitrification superimpose at a distance of 90–150 m from the landfill. As discussed previously, overlap between the iron-reducing and the manganese-reducing zone is likely. Correspondingly, manganese reduction and denitrification have been shown to take place simultaneously in sediments (see review by ref 18). Methanogenesis and sulfate reduction may both take place under some conditions depending on availability of electron donors and substrate. Coexistence of methane and sulfide may in these cases be explained by the presence of microenvironments in the aquifer where different processes take place. Another possible explanation is the transport of compounds from an upgradient methanogenic zone to the sulfate-reducing zone. However, these explanations are extremely hard to debate solely on groundwater sample characteristics. Bioassays performed on sediment from the redox zones in the Vejen Landfill leachate plume (28) revealed that in many cases the microbial potential to perform the identified redox processes could also be identified outside the actual redox zones, e.g., sediment from the iron-reducing zone had a potential to perform iron reduction and nitrate reduction, although not simultaneously. This supports the existence of overlapping zones and the various transition zones identified in the Grindsted Landfill leachate plume.

The detailed characterization of groundwater samples from the Grindsted Landfill leachate plume has revealed a sequence of redox zones in the plume. The slight differences between the two transects, the transition zones identified, and the inherent limitations of the employed redox criteria as discussed in this paper however show that the redox zones as the chemical frame work for the attenuation processes in the plume should be viewed as indications of the governing redox processes and without sharply defined boundaries.

However, this investigation clearly shows that pollutants from the landfill will migrate through many different redox environments in the plume. Thus, pollutant attenuation in leachate plumes must be viewed in due respect of the different redox environments.

Acknowledgments

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Air Sparging

by R.L. Johnson, P.C. Johnson, D.B. McWhorter, R.E. Hinchee, and I. Goodman

Introduction

In situ air sparging (IAS) is a technique in which air is injected into water saturated zones for the purpose of removing organic contaminants by a combination of volatilization and aerobic biodegradation processes. It is typically used in conjunction with soil vapor extraction (SVE) to eliminate the off-site migration of vapors. Its use for the remediation of gasoline and chlorinated solvent spill sites has been reported. Air sparging has broad appeal because, like SVE, it is relatively simple to implement and capital costs are modest. However, like most subsurface remediation activities, in situ air sparging relies on the interactions between complex physical, chemical, and biological processes, many of which are not well understood.

This paper discusses several issues related to in situ air sparging. First, the current state of the practice of air sparging is described. Second, physical and biological processes that control the performance of IAS systems are discussed. Finally, a review of design criteria for implementing IAS is presented. The material that follows is primarily a review and critical evaluation of currently available literature on this subject. Because the available information is somewhat limited, the authors have drawn upon their own experience to provide a context for interpreting reported IAS performance data. This overview is not intended to be exhaustive, nor is it the final word on in situ air sparging. Instead, it is anticipated that this paper will raise

Abstract

In situ air sparging (IAS) is becoming a widely used technology for remediating sites contaminated by volatile organic materials such as petroleum hydrocarbons. Published data indicate that the injection of air into subsurface water saturated areas coupled with soil vapor extraction (SVE) can increase removal rates in comparison to SVE alone for cases where hydrocarbons are distributed within the water saturated zone. However, the technology is still in its infancy and has not been subject to adequate research, nor have adequate monitoring methods been employed or even developed. Consequently, most IAS applications are designed, operated, and monitored based upon the experience of the individual practitioner.

The use of in situ air sparging poses risks not generally associated with most practiced remedial technologies: air injection can enhance the undesirable off-site migration of vapors and ground water contamination plumes. Migration of previously immobile liquid hydrocarbons can also be induced. Thus, there is an added incentive to fully understand this technology prior to application.

This overview of the current state of the practice of air sparging is a review of available published literature, consultation with practitioners, a range of unpublished data reports, as well as theoretical considerations. Potential strengths and weaknesses of the technology are discussed and recommendations for future investigations are given.

research, discussion, and writing on this increasingly popular remediation approach.

State of the Practice of In Situ Air Sparging

The goal of an IAS system is to remove volatile and/or aerobically biodegradable hydrocarbons from both ground water and unsaturated subsurface zones. To accomplish this, air sparging systems commonly consist of the following components (Figure 1): (1) air injection well(s); (2) an air compressor; (3) air extraction well(s); (4) a vacuum pump; (5) associated piping and valving for air movement systems; and (6) an off-gas treatment system (e.g., activated carbon, combustion). Depending upon characteristics of the subsurface and the IAS/SVE system, practitioners may select injection air rates ranging from a few to several standard cubic feet per minute (scfm) per well (Table 1). Air injection wells are generally placed a few meters below the water table in the hope of inducing lateral spreading of air away from the injection well. To date, most decisions on injection well placement and flow rates have been based on operator experience.

As air moves up through the ground water zone, contaminants partition into the gas phase and are swept out of the ground water zone to the vadose zone. At the same time, oxygen in the injected sparge air partitions into the ground water. This oxygen may then serve to stimulate the aerobic microbial degradation of contaminants. To prevent the unintended migration of contaminant vapors, sparging systems are integrated with an SVE system at most sites. In general, the rate of air removal by the SVE-system should be substantially greater than the injection rate for the IAS system. Current practice among some practitioners is to adjust rates empirically to ensure overall negative air pressure throughout the remediation zone.

Physical and Biological Processes that Control IAS

Conceptual Model of Injected Air Flow in the Saturated Zone

The flow of air from an injection well toward the vadose zone is the central feature of IAS operations. For the purposes of this discussion, the flow of injected air through the well screen and through the saturated zone toward the unsaturated zone is best discussed in terms of a conceptual model. In this section such a model is presented as the context for discussing processes important to IAS as well as those important for monitoring field performance of IAS systems.

When air is injected into a well, standing water in the well bore is displaced downward and through the well screen until the air/water interface reaches the top of the well screen. The minimum air pressure required for this displacement is the hydrostatic pressure P_H corresponding to the water column height that is displaced:

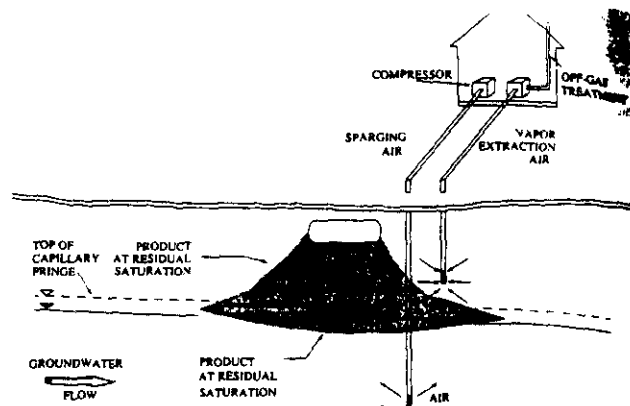


Figure 1. Schematic drawing showing the components of an in situ air sparging/soil vapor extraction system.

$$P_H = r_w g (L_s - L_{gw})$$

where:

- r_w = the density of water ($= 1000 \text{ kg/m}^3$)
- g = the acceleration due to gravity ($= 9.8 \text{ m/s}^2$)
- L_s = the depth to the top of screen (m)
- L_{gw} = the depth to ground water (m).

For the injected air to penetrate the aquifer, air pressure in excess of the hydrostatic pressure is required. This excess air pressure is commonly known as the "air-entry pressure" for the formation, P_{entry} . It is the minimum capillary pressure (air pressure minus water pressure) necessary to induce air to flow into a saturated porous medium. Air-entry pressures (expressed as equivalent water "heads") can range from a few centimeters for coarse sandy soils to several meters in low-permeability clayey soils. If specialized diffuser screens are used to enhance air distribution, then the minimum bubbling pressure for the diffuser ($P_{diffuser}$) must be overcome for air to enter the formation.

As injected air enters the saturated aquifer, it rises due to both its buoyancy in water and the pressure gradient induced by the vapor extraction system. As water is necessarily displaced when air is injected into the formation, a slight rise in the ground water level in the vicinity of the injection well is likely to be observed. However, contrary to some published reports, it is unlikely that the air injection by itself will result in a sustained mound of water within the porous medium. It is more likely that any observed sustained mounding is a result of the vapor extraction system, which can cause sustained ground water upwelling. The water level changes observed in monitoring wells may also be the result of preferential air movement to the wells, and not a reflection of conditions in the formation.

It is virtually impossible to predict the flow path that air channels will take between the injection point and the vadose zone for real field settings. It is well known that water displacement by the invasion of air is remarkably sensitive to even subtle changes in soil structure. Under experimental conditions (Ji et al. 1993; Johnson 1993), the formation of individual air channels occurring at spacings on the order of centimeters, or greater, have been observed. The equivalent diameter of individual



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= 9.8 m/s²)
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air channels is estimated to be, at most, on the order of a few grain diameters. It is important to note that, for realistic scenarios, the air occupying the individual air channels is continuous; in no sense does air flow occur as a sequence of rising bubbles (Figure 2).

Small variations in permeability, or soil structure, at the scale of even a few grain diameters will cause air channels to form. Larger scale heterogeneity, such as stratification, also affects air flow patterns, as demonstrated by Ji et al. (1993) in laboratory visualization studies. For example, if air is injected into a stratum lying below a more fine-grained (higher air-entry pressure) water saturated zone, then the injected air will accumulate beneath the finer grained stratum and form a thin, relatively continuous "bubble" as shown in Figure 3. Lateral spreading of the air will continue until the pressure within the bubble exceeds the air-entry pressure of the finer grained stratum, or until a vertical pathway, such as a monitoring well or fracture, is reached. Field observation of bubbles in monitoring wells has often been interpreted as an indication of air distribution within the medium, while it is more likely an indication of the type of flow described above. It is important to note that flow of this type will also likely cause enhanced transport of hydrocarbons away from the source area.

Processes Controlling the Removal of Contaminants

Air sparging depends on two basic processes for contaminant removal: volatilization and aerobic biodegradation. Similar factors control both processes. This section compares these processes for several areas of the subsurface, including the air flow channels, saturated soils surrounding the air channels, capillary fringe, and vadose zone. Within these areas, contaminants targeted for remediation may be dissolved in the ground water, be adsorbed onto soils, or occur as globules of immiscible non-aqueous-phase liquid (NAPL).

Volatilization

For contaminants initially located within the air channels, volatilization due to air sparging is analogous to vadose zone SVE, and similar removal rates and remedial efficiencies can be anticipated. Where NAPL is in contact with an air channel, contaminants will volatilize by direct evaporation from the NAPL surface. Given the postulated conceptual flow model, the greater contaminant mass will likely be located beyond the air channels in water saturated zones. Removal of this mass will depend upon diffusive transport to the air-water interface, which is inherently a slow process. This analysis leads to the conclusion that the effectiveness of air sparging could be limited, unless the air flow also induces some degree of mixing within the water saturated zone. The injected air eventually moves across the capillary fringe and into the vadose zone, unless it intercepts some preferential conduit to the ground surface, such as a monitoring well. As a result, this might enhance remediation of capillary fringe soils not otherwise affected by SVE, or may simply accelerate remediation

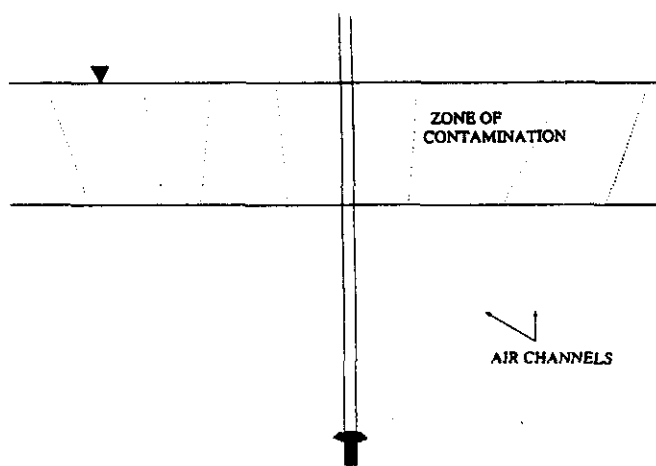


Figure 2. Schematic drawing showing air channels formed during in situ air sparging.

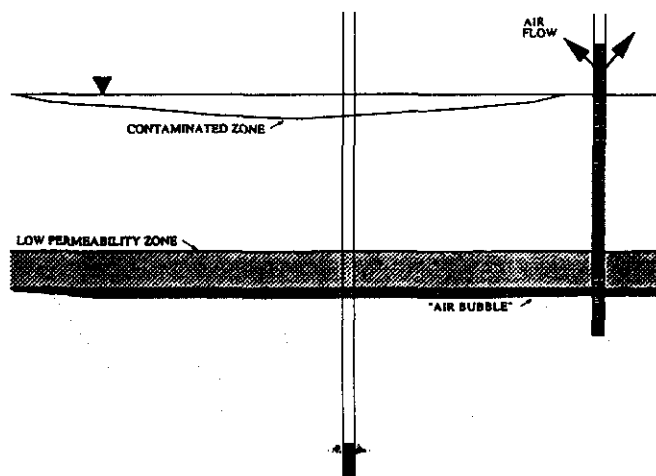


Figure 3. Schematic drawing showing air "short-circuiting" through a monitoring well and around the zone of contamination.

of the portion of the capillary fringe that SVE would treat more slowly. Neither possibility can be confirmed with available data. IAS has been used at sites where hydrocarbon removal by conventional soil vapor extraction has reached "asymptotic" levels. Some studies have reported an initial, short-term increase in hydrocarbon removal rates when air sparging is initiated. However, it should be noted that the cumulative mass of contaminant removed by volatilization during this phase of the remediation is typically a small fraction of the total amount removed over the entire duration of remediation.

Biodegradation

Many compounds in hydrocarbon fuels will biodegrade aerobically; at most fuel-contaminated sites, oxygen is the primary factor limiting biodegradation rates (other potentially limiting factors will not be considered here). IAS is one of a number of methods for delivering oxygen to the saturated zone, and it therefore has the potential to stimulate aerobic biodegradation. Conventional in situ oxygen delivery processes have either used the injection of oxygenated water, water

to increase subsurface oxygen levels. Air-saturated water contains 8 to 10 mg-O₂/L. Oxygen-saturated water increases this level to about 40 mg-O₂/L, and as much as 500 mg-O₂/L can be supplied by water containing hydrogen peroxide. The difficulty in injecting oxygenated water is the relatively high oxygen demand of aerobic hydrocarbon degradation. Between 3 and 3.5 g of O₂ per gram of hydrocarbon is required for complete mineralization, and, at concentrations typical of NAPL-contaminated soils, hundreds or even thousands of pore volumes of water may be required to treat aquifer soils. Practitioners are now beginning to realize the advantage of supplying oxygen through vapor transport. For example, the practice of "bioventing" takes advantage of this feature for vadose zone soils, but it does little to supply significant oxygen to aquifer solids unless the solids are dewatered first. From this perspective, IAS has the potential to be an oxygen delivery method that is at least competitive with current practices.

As previously discussed, contamination in the air channels will be treated much like soils undergoing vapor extraction in the vadose zone, and current experience with bioventing should be applicable. In these channels oxygen will be supplied relatively efficiently and aerobic biodegradation will be stimulated. This may result in the biodegradation of some part of the more volatile fraction and much of the less volatile, higher molecular weight compounds. At fuel-contaminated sites, bioventing of vadose zone soils typically results in biodegradation rates of 2 to 20 mg/kg-d (Hoeppel et al. 1991). Similar rates may be anticipated in the air channels.

Biodegradation of contaminants outside the air channels will be affected by the same mechanisms that control their removal by volatilization. The rate of biodegradation is likely controlled by the rate of oxygen transfer to the ground water, which, as previously stated, is probably limited by diffusion.

Few well-documented air sparging demonstrations have been published. Billings (1991) has applied air sparging to numerous fuel-contaminated sites and, at some, observed concentrations of dissolved hydrocarbons in monitoring wells to decrease in excess of 99 percent in six to 12 months. At other sites, decreases have been less dramatic. Marley et al. (1992) have reported the remediation of a small site where concentrations remained low for a sustained period following IAS shutdown. However, there are few reported cases in which ground water cleanup levels have been achieved and maintained for several years. It also appears that confirmatory soil sampling has been limited at most IAS sites.

Design, Operation, and Monitoring of Air Sparging Systems

In situ air sparging systems should be designed and operated to optimize volatilization and biodegradation

processes and to minimize the probability of adverse consequences, such as off-site migration of vapor or contaminated ground water. As mentioned previously, there is limited design and operation information available in the form of published reports. The guidelines given below, therefore, also include theoretical considerations, empirical results, and practical engineering and economic limitations.

Design Considerations

It is important to recognize that the design of most IAS systems will be based on relatively limited site-specific information. Given this reality and a knowledge of the wide range of behavior that can occur, it is imperative that the potential for flexible operation and system expansion be incorporated into any system design.

Table 1 lists some design specifications for basic air sparging systems and a range of values summarized from published reports. These and other critical design specifications are discussed later on in more detail.

Table 1 Design Parameters for Air Sparging Systems (based on literature values)	
Parameter	Reported Value
<i>Injection Well Specifics</i>	
• screen depth below water table (ft):	16 ¹ , 3 ^{3,5} , 9 ³ , 15-40 ³ , 5 ⁶ , 10-39 ⁸
• screen interval width (ft):	2 ^{1,3,5,8} , 300 ⁴ , 6 ⁶
• number of wells:	14 ¹ , 5 ² , 13 ^{3,5} , 1 ^{4,6,8}
• injection air flow rate (ft ³ /min):	6 ² , 2-6 ^{3,5} , 170-270 ⁴ , 56 ⁶ , 7-16 ⁸ , 3-4 ¹⁰
• injection air pressure (psig):	1-2 ³ , 1-8 ^{3,5} , 3-4 ¹⁰
• operation (pulsed or continuous):	continuous ^{1,2,6,8,9,10} , pulsed ^{3,5}
• other information:	nested injection/extraction wells ^{1,9,10} individual wells ^{2,3,4,5,6,10} horizontal wells ⁴
<i>Vapor Extraction Well Specifics</i>	
• # extraction wells/# injection wells:	8/14 ¹ , 1/1 ^{6,9,10} , 2/13 ^{3,5} , 0/1 ⁸
• extraction flow rate/injection flow rate [ft ³ /min]:	475/30 ² , 580/170 - 580/270 ⁴ , 160/100 ⁶ , 2/1 ¹⁰

1. Brown and Fraxedas 1991
2. Middleton and Hiller 1990
3. Marley et al. 1990
4. Kaback et al. 1991
5. Marley 1991

6. Bohler et al. 1990
7. Wehrle 1990
8. Griffin et al. 1990
9. Ardito and Billings 1990
10. Billings 1991

Air Injection Wells

Air injection wells are usually similar in construction to standard ground water monitoring wells; the main difference is that the screened (perforated) section of

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wells; the main
ted) section of

an air sparging well must be located entirely within the saturated zone. One such construction is depicted in Figure 4. Here the air injection well is placed within a borehole, a relatively permeable packing material surrounds the well screen, and grout seals the annulus above the well screen to inhibit short-circuiting of the injected air. While Figure 4 illustrates a well placed within a borehole, it should be noted that wells may be installed in some soils by driving the casing into the soil. Most published air sparging application summaries report the use of vertical wells (Ardito and Billings 1990; Bohler et al. 1990; Griffin et al. 1990; Marley et al. 1990; Middleton and Hiller 1990; Wehrle 1990; Billings 1991; Brown and Fraxedas 1991; Marley 1991); however, this predominance should be regarded as a reflection of current drilling and well installation procedures rather than an indication that vertical wells offer maximum or unique performance. The use of horizontal wells, which may offer some advantages relative to vertical wells, is reported by Kaback et al. (1991). Other authors report dual vapor extraction/air injection wells constructed by installing separate injection and extraction wells in the same borehole or casing (Ardito and Billings 1990; Billings 1991; Brown and Fraxedas 1991).

The most common material for well construction appears to be PVC, although more heat resistant materials are required if the injected air is warmed too much by the air compressor. Injection well diameters range from 1 to 4 inches; performance is not expected to be affected significantly by changes in well diameter, although as the diameter of the conduit is reduced, the pressure drop due to flow through the piping increases and may become significant. All other factors being equal, economic considerations favor smaller diameter wells (1 to 2 inches), because these are typically less expensive to install and in many cases may be driven into the soil.

Based on the previous discussion concerning the behavior of air injected into an aquifer and the resulting vaporization and biological processes, the well screen location and length should be chosen to maximize the flow of air through the zone of contamination. The top of the well screen, therefore, should always be placed below the lowest suspected level of contamination. This requirement applies equally to vertical and horizontal wells. In relatively homogeneous soils, increasing the depth will tend to expand the zone through which air flows. However, in more heterogeneous and layered soils, increasing the depth beyond the zone of contamination may cause the air flow to circumvent contaminated soils as it seeks the path of least resistance. In either case, water table fluctuations must be considered and the top of the well screen must be placed at a depth where it will not become exposed if the water table drops. Reported well screen length (vertical wells) are 0.5 to 2 m in many cases (Bohler et al. 1990; Marley et al. 1990; Billings 1991; Brown and Fraxedas 1991; Griffin et al. 1990; Marley 1991), and theoretical considerations indicate that there may be little advantage to expanding the screened interval beyond this value.

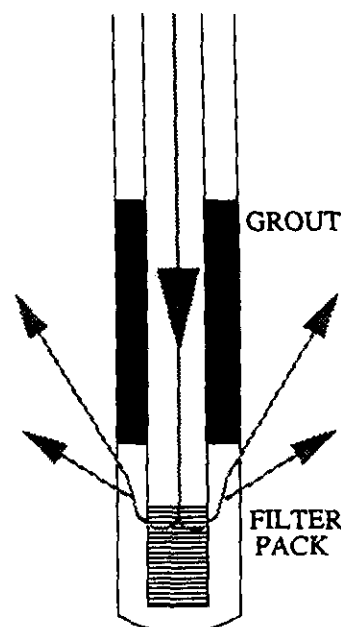


Figure 4. Schematic drawing showing air flow in a well screen and filter pack.

Vapor Extraction Wells

Vapor extraction is typically used in conjunction with air sparging systems in order to remove and treat contaminant vapors liberated by the air sparging process, and to minimize the potential for contaminant vapor migration to nearby structures and conduits. In some cases it may be argued that vapor recovery systems are not necessary: i.e., in remote locations where total potential emission rates are below acceptable levels, or in cases where the injected air flow rate is so low that contaminant vapors are degraded as they pass through the unsaturated zone.

Vapor extraction wells for air sparging applications are usually constructed in the same manner as those used in traditional soil venting applications; vertical wells resemble the air sparging well in Figure 4, with the exception that the screened section of the well must extend at least partially into the unsaturated zone. Horizontal wells or trenches may also be used. Some authors report dual vapor extraction/air sparging wells that incorporate extraction and injection abilities in the same borehole or well casing. This configuration offers obvious economic advantages relative to placing extraction and injection wells in separate boreholes.

At some sites the IAS/SVE system has been designed to remediate soils in both the unsaturated and saturated zones. In such cases the vapor extraction wells should be designed to optimize vapor flow through the contaminated soils above the water table and ensure collection of vapors liberated by air sparging. The reader is referred to Johnson et al. (1990) for some vapor extraction system design considerations. For IAS system designs requiring vapor extraction wells screened near the capillary fringe and water table, it is important to consider ground water level fluctuations when choosing the location of the well screen and screen width.

Well Placement

The number and placement of air injection wells should be chosen to maximize air flow through the contaminated zone. Literature reports often allude to the "radius of influence" or "zone of influence" of an air sparging well; the number of air injection wells is then chosen to ensure that the contaminated zone is encompassed by the zone of influence of the collective system of individual wells. Unfortunately, radius of influence estimates are empirically based, and it is not clear how this quantity should be measured in the field. Some authors claim to have measured it via indirect measurements, such as pressure responses in the unsaturated zone and the bubbling of air in monitoring wells, but the reported evidence is not very defensible. Based on the preceding fundamental discussion of air flow, the zone through which air flows is sensitive to aquifer properties, and a wide range of behavior is possible. Theoretical and experimental analyses of the concept of radius of influence in homogeneous and heterogeneous media are needed to provide a baseline for understanding the spacing and depth of injection of air injection wells.

In the absence of any proven guidelines, it is useful to examine reported injection well spacings: Ardito and Billings (1990) and Billings (1991) seem to prefer to space wells 10 to 20 feet apart, Brown and Fraxedas (1991) appear to have placed wells 50 to 75 feet apart, and 30 to 150 feet spacings are reported in Bohler et al. (1990). It should be noted that these data correspond to vertical well installations, and horizontal wells may prove to be more effective. Theoretical considerations indicate that increasing the number of wells (decreasing the spacing) should increase the rate of remediation in most cases; thus as many wells as possible should be installed, within economic constraints.

The number of vapor extraction wells should be chosen to maximize the recovery of liberated contaminant vapors and to prevent the intrusion of vapors into nearby buildings, conduits, or other enclosed spaces. Table 1 summarizes the relative numbers of extraction and injection wells for some reported applications. Relative to other reported applications, the approach used by Ardito and Billings (1990), Billings (1991), and Brown and Fraxedas (1991) might be regarded as conservative. They utilize dual vapor extraction/air injection well nests; therefore, there is one extraction well for each injection well. These designs are apparently based on the premise that the area of influence of the vapor extraction well will extend beyond the zone where air flow channels emerge from the saturated zone.

Aboveground Components

Given vapor extraction and air injection flow rates (discussed below), one can choose an appropriate blower, compressor, or vacuum pump by finding a unit capable of producing the desired flow rate at an estimated operating pressure or vacuum. The minimum operating pressure for the air injection blower or compressor is equal to the pressure head at the top of the

well screen (2.3 feet below water table equals 1 psig) plus the air entry pressure required to overcome capillary forces. One should be careful to consider potential water table fluctuations when estimating this minimum operating pressure. The operating vacuum for vapor extraction systems can be estimated with simplistic screening model calculations, such as those given by Johnson et al. (1990). Following are other considerations regarding air flow in IAS systems: (1) air injection equipment must produce a contaminant-free vapor stream (many compressors utilize oil for seals) to avoid introducing new contaminants to the aquifer; and (2) safety considerations dictate that air sparging/vapor extraction systems be constructed in such a manner that air injection ceases automatically whenever the vapor extraction system malfunctions.

The use of heated air injection has been reported. The purpose is to heat soils and increase degradation and volatilization rates. Heating probably has limited effectiveness, at least for enhanced biodegradation. The volumetric heat capacity of dry air at standard temperature and pressure is 0.00028 cal/cm^3 , whereas the heat capacity of saturated soils is approximately 0.7 cal/cm^3 . Consequently, at feasible air flow rates and temperature differences, it is not possible to significantly warm soils. For example, an air sparging system injecting 20 scfm of 80 C air into an aquifer at 10 C, 10 feet below the water table affecting a radius of about 20 feet (assuming 12,500 feet^3 of soil uniformly impacted), would result in a maximum temperature rise of approximately 0.06 C per day. This is at or below the level of heating expected from enhanced biodegradation processes. Higher air injection temperatures are possible, but would be detrimental to biodegradation.

Operating Considerations

As previously discussed, increases in air injection flow rate will increase the rate of remediation in most cases. Based on this observation, the air injection system should be operated at the maximum flow rate. However, five other factors limit the rate of air injection:

1. **Mechanical limitations:** Increased flow rates require larger operating pressures and may exceed the capacity of the IAS hardware.
2. **Soil matrix considerations:** As already mentioned, the operating pressure increases as the air injection rate is increased. When this pressure becomes comparable to the overburden of soil above the well, it can cause deformations of the soil matrix or upheaval (fluidization) of the soil above the air injection point. Performance is expected to be best for well-graded medium to coarse sands. This is because less pressure will be required to sustain air injection than required in less permeable soils. In addition, preferential air channeling and poor air distribution are expected to increase significantly as permeability decreases and/or soil heterogeneity increases.
3. **Vapor extraction limitations:** In situations where vapor recovery systems are required, the air injection flow rate must always be less than that of the extraction system flow rate. The extraction system is in

equals 1 psig) overcome capillary forces. This minimum pressure is required for vapor extraction. Plastic piping is given by considerations of injection equipment, vapor stream, and (2) safety. Vapor extraction that air injection

been reported. The degradation has limited biodegradation. The standard temperature increases the heat by 0.7 cal/cm³. Temperature of warm soils, heating 20 scfm set below the about 20 feet (y impacted), use of approximately the level of rate in a process are possible.

air injection in most extraction systems. However, extraction rates require the capacity

mentioned. Air injection becomes complex when the well, it is or upheaval injection point. A well-graded less pressure than required differential air is expected to decrease and

atic where air injection of the extraction system is in

turn limited by characteristics of the vapor extraction blower, or vacuum pump, and the vapor treatment system.

To the extent that remediation is diffusion-limited, increased air flow will serve primarily to increase diffusion gradients (by replacing contaminated or deoxygenated air). At higher air flow rates, a diminishing return may be observed.

If a sparging system is operated to maximize the remediation contribution due to biodegradation rather than volatilization (for example to reduce off-gas treatment costs), high air flow rates may be problematic. With bioventing systems in the unsaturated zone it has been found that lower air flow rates will enhance biodegradation while minimizing volatilization (Miller et al. 1991; Dupont et al. 1991).

Table 1 contains a summary of relative vapor extraction air injection flow rates reported in the literature. Most reported air injection flow rates are less than 10 scfm per injection well.

There are at least three distinct approaches to operating IAS systems. These can be referred to as "staged," "continuous," and "pulsed" operating strategies. In the staged approach the unsaturated soil zone is remediated first, followed by air sparging. At this time there appears to be no benefit to operating in this fashion, unless the goal is to quantify the relative contribution of air sparging to the overall remediation. Continuous and pulsed systems are differentiated by whether or not the air injection is continuous or intermittent. The available data are too limited to determine which approach is best. If mass transfer limitations prove to govern air sparging system behavior, continuous operation will probably be the preferred option. Should the pulsing of the air injection flow rate enhance mixing in the subsurface, a properly timed pulsed operation could deliver enhanced performance.

Health, safety, and compliance issues will also affect the operating conditions of IAS/SVE systems. For example, discharge of extracted vapors must be in compliance with local air discharge standards. This may require the use of off-gas treatment equipment such as carbon beds or thermal or chemical oxidizers.

Monitoring Considerations

Monitoring data can be used to assess the performance of current operating conditions, to help determine if system adjustments or expansions are necessary, and to determine if off-site migration of contaminant vapors and contaminated ground water is occurring. Table 2 lists a number of items that can be monitored. The aboveground system performance items listed in Table 2 (flow rate, concentration, composition) can be used to estimate the net rate of removal due to volatilization. In some cases it may also be used to quantify the rate of biodegradation induced by air sparging (based on proper interpretation of O₂ and CO₂ data).

In situ response data (e.g., pressure, air flow, water quality) are often puzzling and subject to a wide range of interpretations concerning validity and meaning. For example, consider the case where a monitoring well

Table 2
Potential System Monitoring Requirements

Parameter	How Measured
Aboveground System Performance	
• extraction well flow rate:	flowmeter (rotameter, orifice plate, etc.)
• injection well flow rate:	flowmeter (rotameter, orifice plate, etc.)
• extraction well vacuum:	vacuum gauge or manometer
• injection well pressure:	pressure gauge or manometer
• extraction gas concentration:	flame ionization detector (FID) or explosimeter
• extraction well composition*:	gas chromatography with FID
• respiratory gas concentrations*:	electrochemical cell (oxygen) infrared detector (carbon dioxide)
In Situ Response	
• contaminant levels in soil:	analyze soil sample by appropriate method
• soil gas concentrations:	FID or explosimeter**
• soil gas composition*:	gas chromatography with FID**
• respiratory gas concentrations:	electrochemical cell (oxygen) infrared detector (carbon dioxide)
• soil gas pressure/vacuum:	pressure/vacuum gauge or manometer**
• ground water elevation:	pressure transducer or tape in monitoring well
• contaminant levels in ground water:	analyze ground water sample by appropriate method
• dissolved oxygen levels:	analyze ground water sample

*includes compositional analyses of hydrocarbon (boiling point fractionation or individual species).

**requires vadose monitoring installations or soil gas probes.

intersects a large subsurface "air bubble" (formed in response to stratified soil conditions). Air will bubble up through water in the monitoring well, thereby causing contaminant concentrations in the well water to be lower, and dissolved oxygen levels higher, than concentrations in the surrounding aquifer. Other equally likely scenarios lead to the conclusion that monitoring well samples analyzed during operation of an air sparging system will always be suspect. It is recommended, therefore, that ground water samples collected for the purpose of assessing remediation only be obtained weeks or months after system shut-down. Ground water samples can also be collected utilizing driven devices, or by means other than a conventional monitoring well. It does not appear that monitoring wells are useful in determining ground water oxygen concentrations. As with any in situ remediation technique, soil sampling before and after treatment must be done to confirm effectiveness. This is particularly true with IAS, because conventional monitoring well data are suspect.

Soil gas pressure/vacuum and concentration/composition analyses are relatively reliable indicators of condi-

tions in the vicinity of the monitoring point. These can be collected with the use of permanent vadose zone monitoring installations (Johnson et al. 1990) or driven soil gas probes. In most cases, a measurable vacuum is interpreted as an indication that the monitoring point lies within a zone where vapors are flowing toward the vapor extraction well(s). Unfortunately, in heterogeneous systems, the relationship between vacuum and air velocity is not straightforward, and it may be necessary to have some more direct measurement of velocity if remediation effectiveness is to be predicted. Finally, vadose monitoring locations should be placed near any buildings or conduits if there is concern over the potential migration of contaminants to these locations.

Summary

In situ air sparging systems are more frequently being proposed and installed for remediating aquifers contaminated with volatile organic compounds. The rapid, widespread application of this technology is occurring because it is relatively simple and cost-effective to implement, and because potential risks can be overcome if systems are operated properly. However, for the following reasons, interpretation of IAS performance data is quite difficult and misinterpretation is quite common:

1. The physics of air movement in saturated porous media are not widely understood. Nearly all published reports incorrectly show air movement occurring as bubbles. This will rarely be the case; air flow will almost always occur in small continuous air channels.
2. Air movement within the saturated zone is extremely sensitive to formation structure. Small variations in permeability may control the air pathways within the medium. In this manner, large portions of the targeted remediation zone may be bypassed by the sparge air. The movement of air within the formation is difficult to predict and to monitor.
3. Monitoring of IAS performance is most commonly accomplished using conventional monitoring wells. Unfortunately, the design of these wells often adversely affects the data obtained from them. For example, if sparge air enters the monitoring well, then contaminant and oxygen concentrations within the well may not reflect those concentrations in the formation due to sparging within the well. New monitoring techniques must be developed to allow IAS performance to be effectively monitored.

The effectiveness of IAS in remediating ground water and aquifer solids in the saturated zone is not understood. If the process is diffusion limited, and saturated zone remediation is primarily to air flow channels, most of the remedial benefits of IAS are likely to occur in the capillary fringe and vadose zone. To address this issue, future studies need to focus on mass transfer and remedial processes in the saturated zone. Total removal data and monitoring well data as typically collected do not address this important issue.

Despite these problems, in situ air sparging has potential as a remediation tool, when applied in a safe manner and when its limitations are understood. Given its increasing use, it is essential that the technique be examined in detail so that its strengths and weaknesses can be better understood.

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Biographical Sketches

Dr. Richard L. Johnson is an associate professor in the Department of Environmental Science and Engineering at the Oregon Graduate Institute (P.O. Box 91000, Portland, OR 97291). He is also the director of the OGI Center for Groundwater Research and the OGI Large Experimental Aquifer Program.

Dr. Paul C. Johnson is a senior research engineer at Shell Development Co. (Westhollow Research Center, P.O. Box 1380, Houston, TX 77251).

Dr. David B. McWhorter is a professor in the Department of Agricultural and Chemical Engineering at Colorado State University (Engineering Research Center, Foothills Campus, Fort Collins, CO 80523).

Dr. Robert Hinchee is a research leader in the Environmental Technology Department at Battelle Memorial Institute (505 King Ave., Columbus, OH 43201).

Iris Goodman is an environmental scientist with the Advanced Monitoring Systems Division at the U.S. EPA Environmental Monitoring Systems Laboratory (P.O. Box 93478, Las Vegas, NV 89192).

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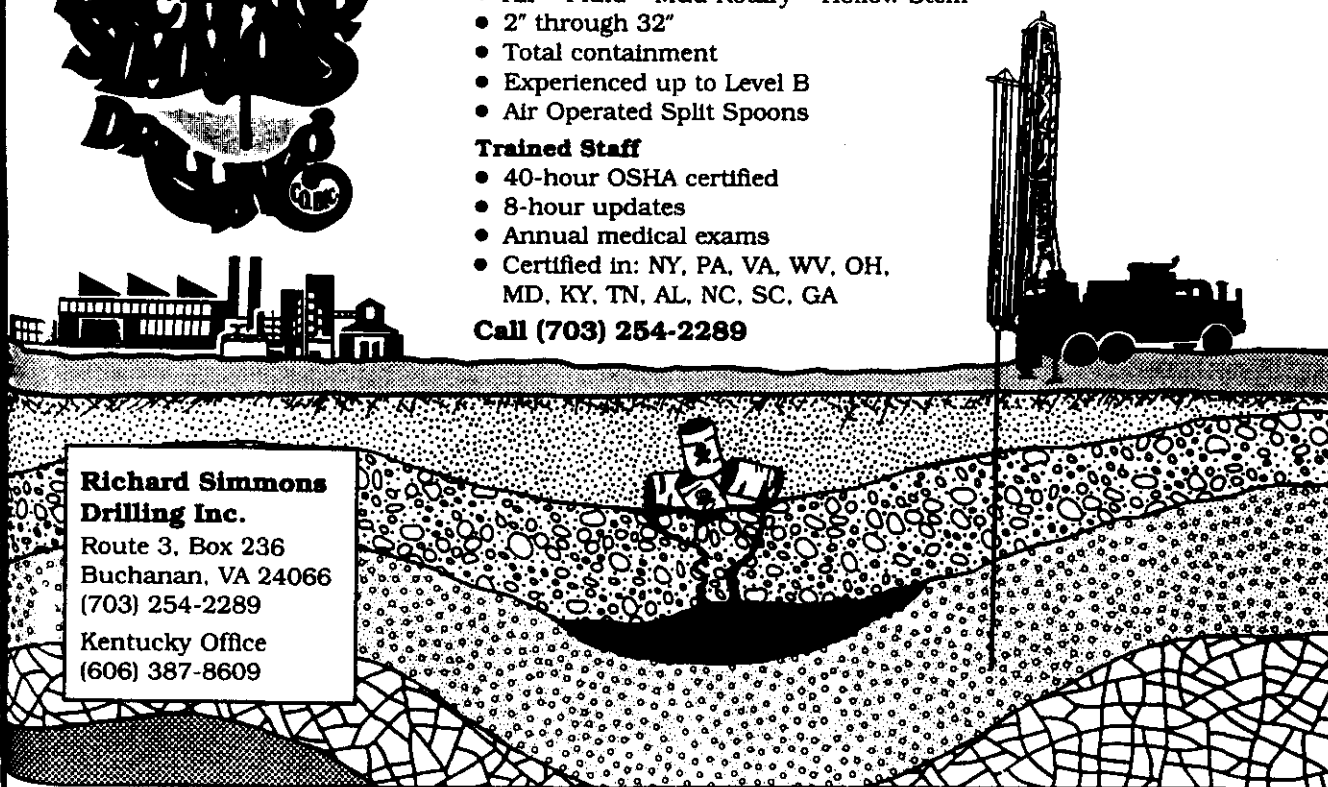
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Modelling and mapping are critical, while air sparging and soil vapor extraction have become strong allies

Maureen C. Leahy and Richard A. Brown
Groundwater Technology, Inc.

As it is now practiced, bioremediation involves stimulating naturally occurring bacteria to degrade hazardous waste in soils and groundwater. While it can be employed as a standalone treatment method, today, bioremediation is seen more as part of an integrated treatment system. Combined with different technologies – particularly air sparging and bioventing – it can help achieve target cleanup goals at the lowest possible cost.

While bioremediation cannot handle metals, and some chlorinated organics still elude it, the technology can destroy many hazardous compounds, including some that resist other forms of treatment. Microbial treatment is more expensive than such techniques as soil

contaminants have been found in the environment. Some chemicals, such as chlorinated solvents and heavy petroleum products, are more resistant to microbes, but can still be degraded by specialized bacterial systems.

Today's commercial bioremediation processes use naturally occurring aerobic bacteria to degrade petroleum and hazardous wastes by oxidizing these compounds. Through a sequence of metabolic steps, the bacteria oxidize the carbon-containing contaminants to carbon dioxide and water, while incorporating some of the carbon into new biomass growth (Figure 2, p. 110).

All essential growth factors for bacterial metabolism must be present for efficient bioremediation. These include



Groundwater Technology

BIOREMEDIATION:

vapor extraction, but cheaper – in many cases – than offsite treatment and faster than many other remediation methods. It can also be used as a “polishing” treatment, to further reduce contaminant levels after another type of treatment has been used.

How it works

Bacteria naturally present in the environment (box, p. 110) are capable of degrading a wide variety of organic contaminants (Figure 1). The vast majority of contaminants that have been treated by bioremediation to date are petroleum derivatives including fuels; petroleum solvents such as acetone and ketones, and polyaromatic hydrocarbons (PAHs) found in coal tars and creosotes [1]. Many naturally occurring strains of bacteria capable of degrading specific

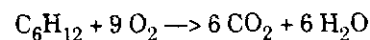
elements such as carbon, oxygen, hydrogen, nitrogen and phosphorus. After carbon, the most important growth factor is oxygen, which acts as an electron acceptor and energy source to drive metabolism of the contaminants.

The other major elements, nitrogen and phosphorus, provide materials for developing biomass, as do essential elements such as sulfur, calcium, potassium and magnesium. With the exception of oxygen, these factors should be provided in direct proportion to the concentrations of these elements in bacterial biomass.

The ratio of degradable carbon to major nutrients in biomass is generally 100 parts carbon to 10 parts nitrogen to 1 part phosphorus. Since biomass is often consumed by the next generation of bacteria, the requirements may be

closer to 300 parts carbon to 10 parts nitrogen to 1 part phosphorus.

Oxygen must be provided in a much higher ratio than that found in biomass, because it drives the metabolic process and is given off in the form of CO₂. Benzene, for example, is biodegraded according to the formula:



In general, aerobic biodegradation of petroleum hydrocarbons requires at least 3 to 3.5 lb of oxygen for every 1 lb of hydrocarbons. One pound of oxygen can be supplied by 60 ft³ of air.

Factors governing success

Assuming that a hazardous waste is biodegradable and appropriate bacteria are present in a given site's soil and groundwater, many other factors –



Aboveground bioremediation is shown here in action, treating petroleum-contaminated soil at a Rotterdam refinery. The onsite technique places contaminated soil in 'biopiles,' or engineered treatment cells. Manifolder piping brings oxygen and nutrients to

each cell, and allows for hookup to process control equipment and monitors. Protective sheeting prevents runoff and traps heat during cold months. The technique avoids transportation and liability, and is expected to cost half as much as offsite treatment

OPTIMIZING RESULTS

some controllable, others not - determine how easily bioremediation can be applied and how effective it will be. A thorough site assessment can save con-

siderable money in the long run (box, p. 111). When bioremediation is being considered for a particular site, the following factors must be evaluated:

Distribution and amount of contaminant

Contaminants may be found in any of four distribution phases:

1. Adsorbed to saturated or unsaturated soils
2. Dissolved in the groundwater
3. As liquid floating on groundwater or settling on a confining geologic layer
4. As vapors.

For example (Figure 3, p. 111), if 10,000 gal of gasoline are spilled in a medium- to fine-grained sand, and depth to groundwater is 15 ft, the gasoline will settle into four distinct phases: 60% will be adsorbed to soil, and 35-40% will float as a liquid on top of groundwater. Only 1-3% will dissolve into groundwater, and less than 1% will volatilize.

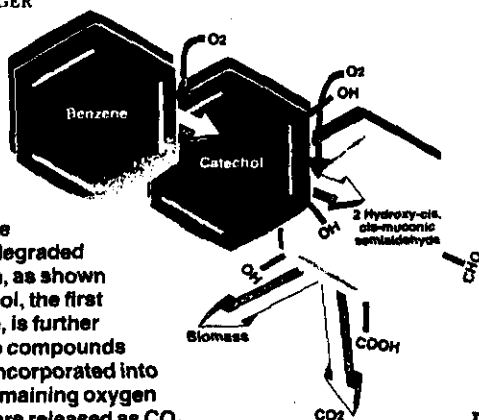
Along with evaluating the phase dis-

BIODEGRADABILITY OF ORGANIC HAZARDOUS WASTE		
READILY DEGRADABLE	MODERATELY DEGRADABLE	HARD TO DEGRADE
Gasoline Jet Fuel Diesel Fuel Toluene Benzene Isopropyl Alcohol Methanol Acetone Ketones Phenols Acrylonitrile	#6 Oil Crude Oil Lubricating Oils Coal Tars Creosotes Pentachlorophenol Nitrobenzene Aniline Long-chain aliphatics Phthalates	TCE PCE Vinyl Chloride PCBs DDT Chlordane Heptachlor

FIGURE 1. Although it can't yet degrade all organic substances, bioremediation is handling increasingly challenging pollutants

FIGURE 2.

Benzene, toluene and other substituted aromatics are aerobically degraded via oxidation, as shown here. Catechol, the first intermediate, is further oxidized into compounds that can be incorporated into biomass. Remaining oxygen and carbon are released as CO₂.



WHICH BACTERIA DEGRADE ORGANIC HAZARDOUS WASTE?

COMPOUNDS	MICROBE
Aliphatics hydrocarbons	<i>Pseudomonas</i> sp. <i>Acinetobacter</i> sp. <i>Mycobacterium</i> sp. <i>Candida</i> sp. <i>Anthrobacter</i> sp.
Chlorinated solvents	<i>Methylobacter</i> sp. <i>Methylococcus</i> sp. (methane users) <i>Pseudomonas putida</i> (phenol users)
Aromatics	<i>Mycobacterium</i> sp.
Chlorobenzoates	<i>Pseudomonas</i> sp. <i>Nocardia</i> sp. <i>Anthrobacter</i> sp.

This list represents a sample of the microorganisms [9,10,15,16, and 17] capable of biodegrading hazardous waste. Both in situ and above ground bioremediation rely on multiple strains of naturally occurring bacteria that act together to complete degradation. Their growth is stimulated by the presence of organic hazardous waste, and can be limited in the environment by the availability of O₂ and nutrients.

Using a particular strain of bacteria to degrade hazardous waste in the environment is difficult. Naturally occurring bacteria have a competitive advantage because they are adapted to site specific conditions. Microorganisms cultivated in a laboratory cannot compete with indigenous organisms.

Today, the practical focus of bioremediation in the laboratory is not to identify which naturally occurring bacteria can degrade hazardous waste, but rather what conditions in the soil or water promote the most efficient biodegradation and how to achieve and maintain those conditions. □

tribution of contaminants, it is essential to map the vertical and horizontal extent of the contaminants. These evaluations provide vital information to:

- Determine the effect that one phase may have on another

For example, adsorbed-phase contamination may continuously leach into the groundwater to create a continuous, dissolved-phase problem, despite ongoing dissolved-phase treatment.

- Select a primary treatment approach to address most of the hazardous waste at the site

The primary treatment approach is determined by the type of contamination, its distribution in the subsurface, site conditions and closure goals.

- Identify secondary treatment systems to address the other phases

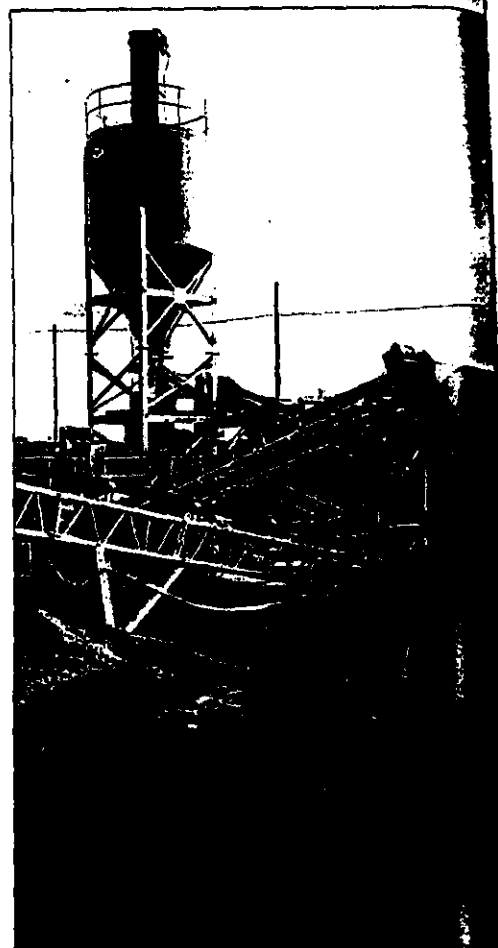
While resources need to focus on the mass of contamination, it is necessary to address all phases because of the relationship between them. Treating the dissolved phase alone, for example, will never restore the site.

- Explore the opportunity for synergy between the various treatment systems to improve efficiency and reduce cost

For example, a soil vapor-extraction system, which is primarily designed to remove contaminants by volatilization, also delivers oxygen to enhance biological degradation. Similarly, groundwater extraction systems not only contain contamination on site, but also increase groundwater velocity to enhance nutrient transport.

The sequence in which various components in a given treatment system are applied can have a tremendous bearing on the system's efficiency. Treatability studies are critical. In one case, such a study showed that a system's efficiency could be doubled by air stripping before, rather than after, treatment in a bioreactor. As it turned out, air stripping removed trace amounts of a bactericide that had inhibited the bioreactor's performance.

- Determine the most effective distribution of groundwater extrac-



Soil conditioning speeds aboveground onsite biotreatment. At this Canadian refinery, predominantly clay soils were mechanically shredded and amended with gypsum, allowing treatment to be completed within a year.

tion and liquid-phase recovery wells and injection galleries

These designs are developed using pump tests and computer modeling. For example, pump tests can estimate groundwater velocity and flow direction during groundwater extraction. Computer models then help select optimal locations for extraction wells to control nutrient transport and contaminant movement.

- Determine the effective spacing of soil vapor extraction and air sparging points using pilot tests and computer modeling

Pilot tests assess the permeability of subsurface soil to air flow, and computer modeling uses these data to predict the volume of soil around a sparge or vent point to which sufficient oxygen

will be delivered for bioremediation.

Bioremediation is highly effective at treating both dissolved and adsorbed phase contamination, particularly the latter, which is hard to treat using groundwater recovery. Generally, however, bioremediation cannot remove large masses of liquid-phase contamination. In these cases, it is best used as a polishing technique, once product recovery techniques have been used to remove the liquid.

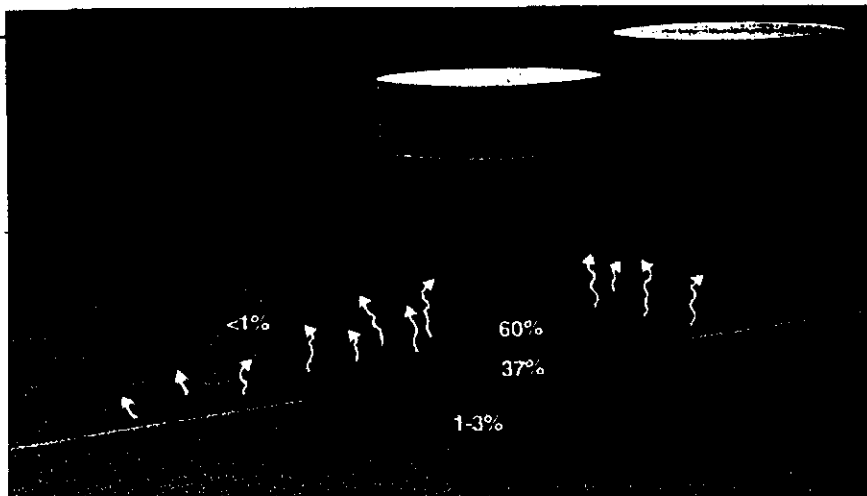
Nature of the Contaminants

The selection of bioremediation and its integration with other technologies requires an understanding of the chemical and physical properties of the contaminants.

•**Solubility** - The more soluble the substance, the more mobile it generally is. Highly soluble contaminants are often amenable to bioremediation; however, groundwater extraction is a more-efficient treatment technology for very soluble compounds.

•**Volatility** - Bioremediation and technologies that remove VOCs by volatilization are often complementary, because both involve the movement of air through the soil or groundwater by soil vapor extraction or air sparging. The air supplies oxygen to support biodegradation, while, at the same time, it physically removes volatile organic compounds (VOCs). How much remediation is due to volatilization and how much to biological activity depends primarily on the volatility of the contaminants and the rate of air flow in the subsurface.

•**Biodegradability** - The biodegradability of contaminants must be determined either through laboratory treatability testing or from reliable published sources. As versatile as naturally occurring microorganisms can be at degrading contaminants, some compounds resist simple aerobic biodegradation. These include many chlorinated compounds, such as trichloroethene (TCE), polychlorinated biphenyls (PCBs), and such high-molecular-weight polyaromatic hydrocarbons (PAHs) as benzo(a)anthracene. While commercially available bioremediation processes cannot effectively treat these contaminants, innovative processes involving specialized microorganisms, both naturally occurring and geneti-



Groundwater Technology

FIGURE 3.

The first step in assessing a site is to determine how pollutants are distributed; typically, they move into several different phases. Shown here is the phase distribution for 10,000 gal of gasoline in medium- to fine-grained sand and groundwater 15 ft below the soil surface: 60% is adsorbed to soil, 35-40% floats on groundwater or settles on a confining geologic layer, 1-3% dissolves into groundwater, and less than 1% becomes vapor

SUBSURFACE MAPPING AND MODELLING

Within the past decade, enormous strides have been made in the development of field sampling techniques and software tools for subsurface mapping and modelling. One field technique is the use of cone penetrometers. These devices are hydraulically driven subsurface probes, capable of sensing the subsurface geology, and collecting water and soil samples at various depths.

The probes provide information relating to soil type and density as well as moisture content. Soil and water samples collected at various depths are used to build a three-dimensional profile of hazardous waste in the subsurface.

The cone penetrometer eliminates the considerable expense of traditional data-collection techniques such as soil boring and drilling. With the exception of samples, little contaminated soil and groundwater is generated that requires offsite disposal. This tool makes it possible to collect more data in less time for lower cost.

Other advancements for mapping subsurface conditions include ground penetrating radar, radio frequency testing, electromagnetic surveying and sonic testing. These technologies are most frequently used in complex hydrogeological conditions where it is necessary to map multiple plumes, geologic confining layers or underground obstructions.

Paralleling these field techniques, new and more-effective software has been developed for data management and analysis. Data acquisition software quickly assembles large volumes of data in formats appropriate for other computerized functions such as 3-D subsurface mapping, contaminant volume calculation, or fate and analysis modelling. □

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Treatment costs (\$ per m ³): Ranges and midpoints				
Incineration		350	975	1,600
100	350	600	Landfill	
50	125	200	Thermal desorption	
125	237.5	350	Soil washing	
40	95	150	Aboveground bioremediation	
0		\$400	\$1,000	\$1,600

cally manipulated, are currently undergoing testing and development.

Site characteristics

Site features that play a key role in determining bioremediation's effectiveness include site geology, hydrogeology, soil permeability, soil moisture content, soil chemistry and temperature.

•**Geology** - If geological impediments such as impassable barriers or strata with varying degrees of permeability exist, then in situ bioremediation of certain areas may not be practical.

•**Hydrogeology** - Understanding the dynamics of fluid flow, as determined by site geology, is also vital for designing transport mechanisms for oxygen and nutrients, especially in the saturated capillary fringe.

•**Soil permeability** - Very fine-grained soils, such as silt or clay, are impermeable, making it difficult to transport nutrients into the matrix. For effective in situ bioremediation, the permeability limit is generally considered to be a hydraulic conductivity of 10^{-4} cm/s.

•**Soil moisture content** - Bacteria live in the film of water surrounding soil particles. The more moist the soil, the more volume is available in which bacteria can live. However, for bioremediation of excavated or unsaturated soils, extremely high moisture content can impede the movement of air through soil and limit the supply of oxygen.

The most economical way to bring oxygen to relatively unsaturated soils is to induce a vacuum or inject pressurized air into the soils. This approach is effective up to a maximum moisture content of about 50% of the soil's moisture-holding capacity. Clay soils may hold up to 60% of their weight in water. Sandy soils can hold up to 25% of their weight in water.

•**Soil chemistry** - There are several concerns relating to soil chemistry. In highly organic soils, contaminants may bind with soil particles, limiting its availability as a food source, and impede biodegradation. Inorganic nutrients have a tendency to bind more tightly to clay soils, slowing nutrient transport.

•**Temperature range** - Extremes of heat and cold render bacteria ineffective. If the matrix is frozen, bacteria

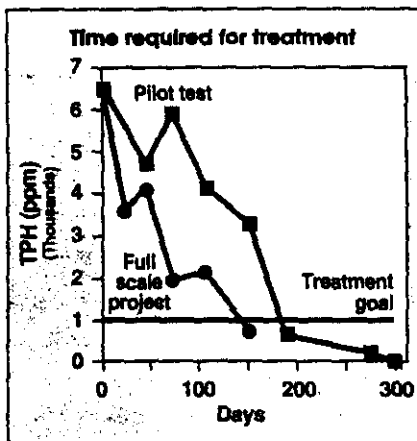


FIGURE 4. Above ground biotreatment reduces total petroleum hydrocarbon (TPH) concentrations rapidly, at first, and then more slowly over time

will be immobilized, and bioremediation will grind to a halt. Most soil bacteria are adapted to the relatively stable, cool temperature - typically, 10 to 15°C - common in most soil subsurfaces. These bacteria will become inactive at temperatures above 30°C. Soil bacterial populations are very complex, however, and some thermophilic bacteria are effective above 70°C.

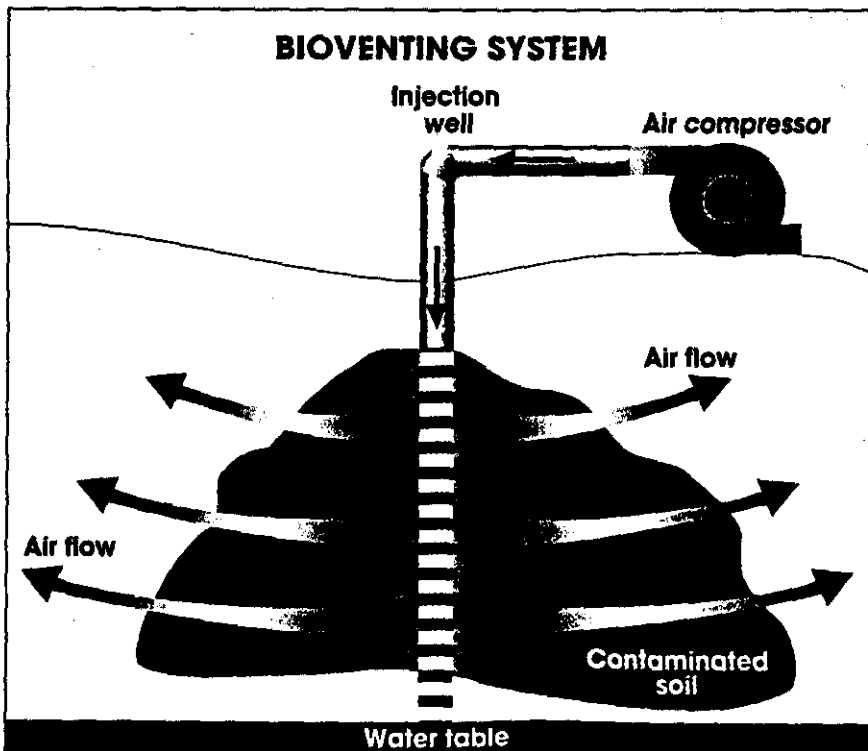
Field tests done by the U.S. Air Force in Alaska indicate that each 10°C increase in temperature brings about a two-fold increase of bacterial activity [2]. Soil temperature can then be raised in situ by injecting hot air or steam, but the benefits of increased temperatures must be weighed against the cost of supplying the heat.

Bacterial activity generates its own heat, and large aboveground soil piles can hold enough microbially-generated heat to support bioremediation well into the winter. In such situations, when the temperature in an individual treatment cell drops below 5°C, energy can be saved by turning off the air supply systems.

Bioremediation in Practice

Onsite bioremediation can be performed either in situ or in aboveground engineered treatment cells if the soils can be excavated. Aboveground treatment can be very rapid (Figure 4), reaching objectives in as little as 90 d for petroleum-contaminated soil [3]. Excavation, however, requires that contaminants be accessible and lie at shallow depths. In contrast, in situ

FIGURE 5. Bioventing, shown below, minimizes the expense of offgas treatment by introducing just enough oxygen to stimulate bioremediation without volatilizing contaminants



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bioremediation can be implemented under buildings and other areas with high activity with little or no impact on daily operations; for deep or widespread contamination, in situ biodegradation is often the most cost-effective approach.

•Aboveground cells

Once the decision has been made to excavate the contaminated soils, the treatment approach is generally decided on the basis of cost. Onsite treatment is usually the least expensive option (Box, p. 111). It also offers additional benefits, since the soil can be used again onsite as fill, eliminating the potential ongoing liabilities associated with removing it and hauling it to offsite landfills.

With aboveground treatment, soil conditioning is usually required to optimize oxygen flow, permeability and homogeneity, moisture content, temperature, pH and inorganic nutrient addition. This is usually accomplished by mechanically shredding the soil, adding an inert support material such as sand, or adding a stabilizing agent such as gypsum. Oxygen can then be supplied to the soils via a network of imbedded piping and the vacuum blower or an air compressor. Nutrients are added as soils are loaded into the cells. The need for soil modification must be evaluated on a site by site basis.

•In situ treatment

As with above-ground remediation, a thorough site assessment is the key to success with in situ treatment. Site information determines the major design parameters, such as spacing of vent, sparge and nutrient injection points, the choice of either horizontal or vertical wells, and the depth to which wells should be screened to ensure that oxygen and nutrients are effectively transported to contaminated soils. Nutrient transport is also contingent on an accurate understanding of the paths that nutrients will follow once injected.

For in situ bioremediation, oxygen delivery is the critical factor. Any of the

following methods, initially developed for other purposes, can be used – some of them can be combined:

•Soil vapor extraction (SVE)

Drawing a vacuum through unsaturated soils promotes the flow of fresh air into the subsurface and supplies oxygen to support bioremediation. The spacing of the extraction points [4] and the size of the vacuum depend on the soil's permeability, which can be easily determined by a pilot test. Since SVE will remove volatile constituents in the course of stimulating bioremediation,



FIGURE 6. Air sparging, when used with soil vapor extraction, creates an oxygen-rich zone in the capillary fringe, removing adsorbed and dissolved-phase contaminants through volatilization and bioremediation. At one plant, this reduced trichloroethylene and other volatiles in groundwater by 90% in 13 months

offgas treatment may be required.

•Biovent systems

A variant of SVE, bioventing (Figure 4) minimizes volatilization – and, with it, the need for expensive offgas treatment – by inducing low volumes of air through unsaturated soils. The ratio of bioremediation to volatilization can be maximized by moving only as much oxygen through the soil as the indigenous bacteria can consume.

An in situ respiration test determines the air volume required by measuring oxygen depletion in soil gas over time, and the corresponding increase in carbon dioxide concentration [5]. During the test, oxygen and carbon dioxide concentrations are monitored throughout the contaminated zone after the subsurface has been thoroughly oxygenated by extracting or injecting air.

These data are then used to calculate the oxidation rate of contaminants in the subsurface, and the overall biological oxygen demand (BOD). During full scale bioremediation, oxygen concentrations are typically kept above 10% (volume) in air, and CO₂ concentrations below 5% (volume) in air.

With SVE, vent points are frequently located in the middle of the contaminated zone to increase volatilization. With bioventing, vent points are placed outside of the contaminated zone to create a longer air flow path, stimulating biodegradation across a wider zone with minimal volatilization.

•Air sparge systems

Air sparging, developed over the past few years, is perhaps this decade's most important advancement in in situ bioremediation and saturated zone treatment. It controls the injection of oxygen into saturated soils and groundwater [6]. The injected air moves through the saturated zone, removing adsorbed and dissolved contaminants while delivering large quantities of oxygen to stimulate bioremediation.

•Sparge-vent systems

If air sparging results in a significant amount of volatilization, then an SVE system can be installed to

collect the contaminant-laden air for offgas treatment, if required. When contamination is present in both saturated and unsaturated soils, then the best solution is to use sparging and SVE simultaneously, to promote biodegradation.

An SVE system is normally designed to collect more air than is injected through a sparge system, thereby creating a slight negative pressure in the unsaturated zone. Therefore, designs for both air sparging and soil vapor extraction systems must be balanced, to minimize equipment requirements and operational costs.

•Sparge barriers

A line of sparge points can be used to form a biological barrier to the down-gradient migration of contaminants [7]. Air injected through sparge points cre-

ates a zone of high biological activity, which functions like a bioreactor in the subsurface. Contaminants are degraded as they move through this zone.

The effectiveness of sparge barriers depends on subsurface permeability and the biodegradability of the contaminants present. Subsurface permeability determines the migration rate of the contaminants through the sparge barrier. In highly permeable soils, where groundwater movement may be as high as 1 ft/d, it will take contaminants at least 80 d to move through a sparge barrier with a 40-ft radius of influence. This distance is usually sufficient to reduce the concentration of a readily degradable contaminant by one or two orders of magnitude.

•Coinjection via sparging

One innovative application of sparging shows much promise. In this technique, other gases are injected along with air to stimulate additional processes or reactions in the subsurface. For example, steam or hot air can be injected to increase temperature to stimulate bacterial activity or volatilization.

Injecting methane gas with air will stimulate methanotrophs, which are bacteria that thrive on methane, and produce an enzyme capable of degrading chlorinated solvents such as trichloroethene (TCE). Other chemically reactive gases such as ozone, which can break down many organic compounds, can be injected either as a pre- or post-treatment step.

•Integration of technologies

The sequences and interactions of in situ bioremediation with other treatment techniques at the site must also be considered when a system is being designed. Sequencing involves knowing when to activate bioremediation in the treatment process. When bioremediation is used to polish hazardous waste down to drinking water levels, it is usually most cost effective to add nutrients only after the mass of contamination has been eliminated.

Other treatment technologies can influence bioremediation. For example, groundwater recovery can lower the groundwater table and expose deeper soils to unsaturated conditions, so that they must be treated differently from saturated soils. It can also increase the groundwater's gradient and velocity



'Air sparging, developed over the past few years, is perhaps the decade's most important advancement in in situ bioremediation and saturated zone treatment'

and influence its flow direction, affecting nutrient transport rates.

The frontiers of bioremediation

New generations of technology, in various stages of development, promise to

Oxygen delivery systems are improving the performance of onsite bioremediation and lowering costs, for both above ground projects, as shown here, and in in situ systems

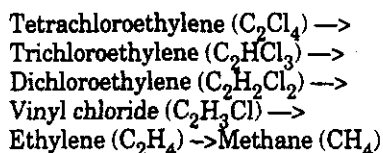
extend the applicability of bioremediation to a broader range of chemicals. Several approaches involve greater use of anaerobic and nitrate reducing bacteria, which have been used successfully in wastewater and sewage treatment, and are capable of degrading chlorinated solvents, PCE, PCBs and other chlorinated aromatics. Laboratory tests are underway on bioengineered bacteria that can degrade those contaminants that continue to resist bioremediation.

Here are some of the advances:

•**Anaerobic bacteria** - These thrive in the complete absence of oxygen. Some anaerobic bacteria are capable of degrading such recalcitrant substances as chlorinated solvents, PCE, PCBs and other chlorinated aromatics. In anaerobic remediation, the electron acceptor is no longer oxygen, but some other chemical such as nitrate, iron, magnesium, or carbon dioxide.

These microbes degrade highly chlorinated contaminants via reductive

dechlorination [8], as shown here:



So far, the key challenge with this technique has been chemically inducing, maintaining, and controlling anaerobic conditions in the subsurface.

• **Chemical oxidation reduction + bioremediation** - Some high-molecular-weight polyaromatic hydrocarbons such as benzo(a)pyrene are highly resistant to biodegradation. However, it is possible to use a brief chemical oxidation reaction to break down a compound into constituents that are readily degradable [9,10]. Chemical oxidants can include ozone or hydrogen peroxide (Fenton's reagent).

• **Cometabolism** - Some contaminants that cannot act as a food or energy source for bacteria can be degraded by cometabolic processes. Cometabolism is a process in which bacteria thriving on one carbon source gratuitously degrade other organic compounds.

Degradation of the contaminant is accomplished via the same enzymes produced by the bacteria to metabolize the food source [11,12]. For example, methanotrophs are a bacterial strain that produce an enzyme to oxidize methane, their food source, that also degrades chlorinated solvents. [13]

Cometabolism requires the addition of the food source to the contamination zone. A complication of these processes, though, is that high concentrations of the food source can inhibit degradation of the contaminant.

• **Aerobic + Anaerobic** - Some compounds such as TCE can be initially dechlorinated and partially degraded by anaerobic bacteria. The products of this reaction can then be degraded rapidly by aerobic processes [14].

Experimental systems that combine aerobic and anaerobic biodegradation for this type of application are now being developed. One approach for in situ applications is to create two remediation zones in the subsurface — an anaerobic zone, and, downgradient, an aerobic zone.

• **Engineered bacteria** - Work is underway at numerous research laborato-

ries to engineer new bacteria to degrade the most stubborn contaminants. There are several problems to overcome prior to commercialization, though. First, systems will have to be developed to transport these bacteria effectively. Second, new bacterial forms will have to be designed to survive outside the laboratory, or conditions within the soil matrix will have to be altered for their survival. The final barrier to the use of bioengineered bacteria for site remediation is the understandable caution of government regulators, who need substantive proof that this

approach presents no risk to either human health or the environment.

In summary, onsite bioremediation has come a long way since its inception in the 1970s. In the 1990s, the major breakthrough has been combining the technique with air sparging, bioventing, and SVE, dramatically reducing cleanup costs and improving performance. As bioremediation becomes more widely used, the future promises to find it in other integrated, multitechnology systems, to break down the most-resistant contaminants. ■

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The authors

Maureen C. Leahy is a senior environmental scientist with Groundwater Technology, Inc., Kennedy Business Park II, 431(F) Hayden Station Road, Windsor, CT. 06095, Tel: 203-688-1151; Fax: 203-688-8239. An expert in bioremediation with over 15 years of experience in microbiology, environmental chemistry, and hazardous waste treatment, she has led several government-sponsored research projects on the biodegradability of chlorinated solvents, jet fuels and coal tars. She received a B.S. in chemistry from Fordham University and M. Phil. and Ph.D. degrees in molecular biophysics and biochemistry from Yale.



Richard A. Brown is vice president of remediation technologies at Groundwater Technology, Inc., 310 Horizon Center Drive, Trenton, NJ, 08691, Tel: 609-587-0300; Fax: 609-587-7908. Since joining the firm in 1988, he has been instrumental in developing innovative, cost cutting onsite soil and groundwater remediation technologies. Over his 12-year career in environmental remediation, he has contributed significantly to improvements in air sparging, soil vapor extraction and bioremediation technologies. He received a B.A. in chemistry from Harvard University, and an M.S. and Ph.D. in organometallic chemistry from Cornell.



The Application of In Situ Air Sparging as an Innovative Soils and Ground Water Remediation Technology

by Michael C. Marley, David J. Hazebrouck, and Matthew T. Walsh

Abstract

Vapor extraction (soil venting) has been demonstrated to be a successful and cost-effective remediation technology for removing VOCs from the vadose (unsaturated) zone. However, in many cases, seasonal water table fluctuations, drawdown associated with pump-and-treat remediation techniques, and spills involving dense, non-aqueous phase liquids (DNAPLS) create contaminated soil below the water table. Vapor extraction alone is not considered to be an optimal remediation technology to address this type of contamination.

An innovative approach to saturated zone remediation is the use of sparging (injection) wells to inject a hydrocarbon-free gaseous medium (typically air) into the saturated zone below the areas of contamination. The contaminants dissolved in the ground water and sorbed onto soil particles partition into the advective air phase, effectively simulating an in situ air-stripping system. The stripped contaminants are transported in the gas phase to the vadose zone, within the radius of influence of a vapor extraction and vapor treatment system.

In situ air sparging is a complex multifluid phase process, which has been applied successfully in Europe since the mid-1980s. To date, site-specific pilot tests have been used to design air-sparging systems. Research is currently underway to develop better engineering design methodologies for the process. Major design parameters to be considered include contaminant type, gas injection pressures and flow rates, site geology, bubble size, injection interval (areal and vertical) and the equipment specifications. Correct design and operation of this technology has been demonstrated to achieve ground water cleanup of VOC contamination to low part-per-billion levels.

Introduction

Accidental releases of volatile organic compounds (VOCs) into the subsurface environment, in the form of petroleum products or industrial solvents, can necessitate costly remediation. Although virtually any form of remediation is expensive, developing a well-planned, cost-effective strategy at the onset of a spill or release can minimize expenses that accumulate during a cleanup project. Removal of the VOC source is usually the primary consideration to ensure effective remediation. Soil contamination that lies beneath and in the vicinity of a leaking underground storage tank or a surface spill is a potential long-term source of hazardous vapors in the vadose zone and dissolved VOCs in ground water. Frequently, contaminated soils exist below the water table when light, non-aqueous phase liquids (i.e., free-phase petroleum products) mound above the saturated zone and are transported vertically in response to seasonal water table fluctuations or drawdown from pumping in nearby ground water/product recovery wells. Dense, non-aqueous phase liquids (DNAPLS) are frequently found in soils below the water table as globules and/or residuals, due to density-driven vertical transport.

A few commercially applicable in situ remediation technologies exist that can be applied as remedial alternatives at VOC spill sites. Generally, no single technique

can accomplish all the objectives of a complete site cleanup. Pump-and-treat methods have been used to recover free-phase product and provide hydraulic control of a migrating dissolved plume, but will have virtually no effect on unsaturated zone soil contamination. Using pump-and-treat methods to remediate VOCs sorbed onto saturated zone soil is considered to have significant limitations (MacKay and Cherry 1985) due mainly to standard pump-and-treat system designs, site-specific soil heterogeneities, contaminant distribution and the kinetic limitations of the mass-removal process. Enhancement of pump-and-treat technology by such techniques as soil flushing is a developing process that requires further review and cost analysis.

In situ biodegradation is another common saturated zone remediation process. This process can be both economical (with respect to existing technologies) and desirable (because it provides for in situ contaminant destruction), but because it deals with a biological process (living organisms), it is sensitive to many environmental and geological parameters which significantly affect its performance and effectiveness. An additional problem with in situ biodegradation involves the difficulties that can occur in attaining regulatory approval, especially where the process requires the introduction of non-indigenous organisms and/or nutrients into the site soils.

Soil-vapor extraction (SVE) has been demonstrated to be a successful and cost-effective remediation technology for removing VOCs from vadose zone soil. This technique involves the controlled application of an air pressure gradient to induce an air flow through soils contaminated with VOCs. As soil gas is drawn toward the vacuum source (vapor extraction well), the equilibrium between the VOC phases (i.e., free-phase product, adsorbed phase, vapor phase, and dissolved phase) is upset, causing enhanced partitioning into the vapor phase. VOCs in the vapor phase are subsequently removed from the subsurface and treated using one of several available off-gas treatment systems. One of the limitations of SVE is that it does not adequately address remediation of contaminated soil below the water table.

A number of techniques have been developed and employed to expand the SVE process to include effective remediation of VOCs in saturated zone soils. Artificial water table drawdown is one approach that may be used to expose contaminated soil in the saturated zone to the advective air phase thereby increasing the efficiency of the soil-venting process. However, in some cases, this is neither a practical nor cost-effective approach. An innovative, alternative approach is the application of in situ air-sparging technology, also referred to as soil/ground water aeration, to inject a hydrocarbon-free gaseous medium (typically air) into the saturated zone below or within the areas of contamination. With air sparging, the VOCs dissolved in ground water and sorbed onto or trapped in the soil, partition into the advective gaseous phase, effectively simulating an in situ, saturated zone air-stripping system. The stripped contaminants are subsequently transported in the air phase to the vadose zone, within the radius of influence of an operating soil-vapor extraction system. The contaminant vapors are drawn through the vadose zone to the vapor extraction well(s) and are treated using a standard vapor extraction off-gas treatment system. A schematic depicting a typical air-sparging system configuration is presented in Figure 1.

Few references exist in the literature concerning the design and/or success of the bench-scale testing or field application of the air-sparging process. Apparently the process was first used as a remediation technology in Germany in the mid 1980s, predominantly to enhance the cleanup of chlorinated solvent-contaminated ground water (Gudemann and Hiller 1988). More recently the technology has been used in the enhanced remediation of gasoline-contaminated saturated zone soil and ground water (Ardito and Billings 1990; Marley 1991; Brown, Herman and Henry 1991, anonymous). In each of these cases, the design of the air-sparging systems has been empirically based. Additionally, articles exist in the literature on the process of injecting air into the saturated zone for the purpose of increasing the dissolved oxygen content of the ground water in order to enhance biological degradation of organic materials (Yaniga, Matson, and Demko 1985).

The authors are unaware of any patents issued on in situ air-sparging technology. Ardito and Billings (1990) mentioned a patent pending on a specific config-

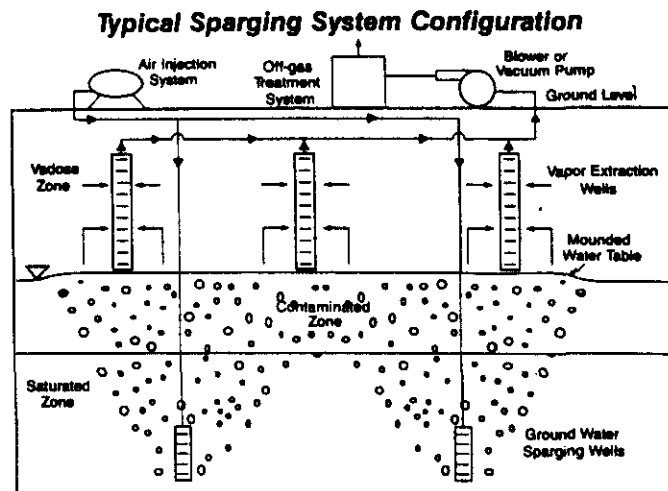


Figure 1. Schematic of a typical air-sparging system configuration.

uration of an air-sparging system. The authors are also aware of a patent application rejection on air sparging due to the existence of prior art. The rejection was based on the presence of air-sparging concepts in the literature describing in situ bioremediation practices from the late 1970s and early 1980s.

The authors have performed air-sparging field pilot tests and implemented full-scale SVE/air-sparging systems on several sites in the northeastern and midwestern United States. Experience developed on these projects has demonstrated that there are numerous important criteria that must be considered when designing, installing, and operating an in situ air-sparging system to ensure effective remediation of saturated zone soil and ground water as well as to preclude displacing and mobilizing potentially hazardous soil-gas vapors, free-phase product or dissolved-phase contaminants in the saturated zone.

The major purpose of this paper is to present to the practicing professional the authors' experiences and conceptual understanding of the application of in situ air-sparging technology. A technical discussion regarding the mathematics of advective air flow in the subsurface and contaminant partitioning dynamics is beyond the scope of this paper; the interested reader is referred to papers by Baehr, Hoag, and Marley (1989), and Baehr (1987) for additional technical information. A technical discussion on the mathematics of the injection of air to enhance petroleum product recovery (a conceptually similar process) can be found in Corey (1986).

Air-Sparging System Design Criteria

As noted previously, the methodologies that are typically applied in the design of air-sparging systems are empirically based. The discussion that follows focuses on a number of the major design parameters that require consideration. The parameters are addressed from both a theoretical and an applied basis. To better understand the complex processes that occur during the operation of a sparging system and to develop an engineered system design process, research at the bench and field pilot-scale is currently underway within

the academic and private sectors.

Bubble Geometry and Gas Channeling

Theoretically, a large number of small bubbles will provide better mass transfer characteristics for the removal of VOCs from the aqueous soil phase than will a smaller number of large bubbles or channels. Air diffusers may be used at the sparging point in order to inject small bubbles into a coarse-grained formation. However, any sand pack around the sparging point should have a grain size that will prevent coalescing of the small bubbles prior to entry into the natural formation. Based on the mechanics of air and water flow in soils, the following assumptions can be made in conceptualizing the in situ air-sparging process:

1. Entry of air into a saturated soil requires pressures greater than the resisting head pressure due to capillary forces. This is known as the "air entry pressure" required to displace water from a saturated soil.

2. Once air entry pressures are overcome, the injected air phase displaces water along paths of least resistance. These paths, or channels, are the result of differences in air entry pressures in the medium caused by micro- and macro-scale heterogeneities.

3. Once a continuous air phase channel is established, it will maintain its integrity as long as the air entry pressure is maintained within the channel.

Contaminant Type

As air sparging is essentially a physical/chemical treatment process (with potential biological enhancements) the compounds that are amenable to remediation through the process are easily identifiable. Generally, those chemicals that are easily removed from contaminated ground water through traditional air-stripping towers are considered optimal for the application of in situ air sparging. Correspondingly, those chemicals that do not strip well have limited remediation potential with sparging. In addition, interactions within the subsurface environment may potentially decrease the effectiveness of the process. Compound sorption onto soil organic material is an example of this. Less obvious are the potential geochemical changes that may occur in the subsurface due to the introduction of gas. The choice of sparging gas to be used at a specific site may be based on these possible interactions. Precipitation of dissolved minerals through changes in oxidation reduction (redox) potential is one of the more obvious potential interactions. The compounds most amenable to air sparging are the lighter petroleum compounds (C_3 - C_{10}) and chlorinated solvents. Less strippable compounds may be remediated with enhancements to the standard sparging process, for example using a combination of air, ozone, and/or hydrogen peroxide as the injected gas to provide increased oxidation potential for semivolatile organic compounds or through potential enhancements to the natural biodegradation process by the addition of oxygen.

Gas Flow Rate

Again, as air sparging is essentially an in situ air-stripping process, it is necessary to provide a sufficient

air-to-water flow ratio to produce the desired contaminant mass removal in a given soil/water volume. In a situation in which ground water control is being exercised, the implications of the hydraulic control system on ground water retention time should be considered in the overall design. Air flow rates that are typically used in the field are in the range of 3 to 10 standard cubic feet per minute (scfm) per sparge point. Pulsing of the air flow into the sparge points is considered, by some, due to potential mass transfer limitations, to provide an energy-efficient and cost-effective approach to remediation. This is a site-specific system design component, and the premise requires further evaluation. Other enhancements in the air delivery process are being investigated that are expected to provide a better distribution of the air channeling to the vadose zone and, hence, provide more effective site remediation.

Gas Injection Pressure

Gas injection pressures are governed by the static water head above the sparge point, the air entry pressure of the saturated soils, and the gas injection operating flow rate. In the design process, the lowest effective air injection pressure will correspond to the pressure required to maintain the minimum gas flow rate that will achieve the desired stripping efficiency. Higher pressures will produce higher air injection flow rates, and may be necessary to provide a more uniform gas channeling distribution in heterogeneous soils due to the range of air entry pressures associated with differing grain size distributions in adjacent soil units. The higher air injection pressures required in fine-grained soils can cause the formation of significant subsurface gas pockets, due to bubble coalescing. A gas pocket is essentially an unsaturated volume that expands from the air-sparging well during the injection process until the pressure within the pocket is sufficient to overcome the vertical air entry pressure of the overlying soils, thereby allowing passage of the injected air to the vadose zone. The pocket expansion will continue until a steady-state condition of air inlet flow to air escape flow is achieved. The vertical channeling of air that occurs will follow those pathways displaying the local, lower air entry pressures. Too high an air injection pressure may create fractures in the sparging well annular seal or along weak joints in the soil, also resulting in a loss of system efficiency.

Site Geology

Site geology is considered the most important design parameter. Air sparging is generally more effective in coarse-grained soil. Coarse soils have lower air entry pressure requirements and provide a medium for more even air distribution, allowing for better mass transfer efficiencies and more effective VOC removal. Fine-grained soils require higher air entry pressures and are more likely to cause the formation of significant gas pockets, which may impede air-sparging effectiveness. The formation of gas pockets also can cause significant lateral displacement of ground water, which, in turn, can cause lateral contaminant displacement and spreading, if ground water control is not used. Heterogeneities

in the soil require the greatest consideration because the potential for non-uniform vertical channeling is significant and has been observed both in the laboratory and in the field. Figure 1 demonstrates the idealized vertical channel distribution (represented as bubbles) expected in a uniform, coarse-grained porous medium, while Figure 2 presents a more typical vertical channel distribution in a soil with non-continuous, fine-grained lenses.

Injection Point Interval

The injection point interval encompasses two topics: (1) the injection well screened interval, and (2) the depth location of the screened interval with respect to the static water table. While the following discussion focuses on vertically screened injection wells, the general concepts outlined are expected to also hold for the application of horizontal injection well systems.

Short screened intervals, on the order of 1 to 3 feet, are generally used in air-sparging wells because most of the air exits through the top of the screened interval, where the pressure head is at a minimum. Use of longer screened intervals does not significantly add to the effectiveness of the process.

In uniform homogeneous soil, injecting at greater depths with respect to the water table tends to increase the radius of influence of an injection point, but also requires higher air pressures at the well to achieve and maintain the gas flow. The existence of significant soil stratifications, as explained previously, tends to enhance lateral displacement of ground water and provide a larger areal extent from which vertical air channels will emanate. One potential problem associated with this situation is the possibility of highly irregular, vertical channeling of gas to the vadose zone that is not likely to provide efficient or effective mass transfer of VOCs from the target contaminated areas.

Radius of Injection Point Influence

The radius of influence of a sparging well can be highly variable, especially in heterogeneous or stratified soils. In coarser soils in which vertical channel distribution is more controllable and predictable, the injected air tends to follow an almost parabolic path to the vadose zone. Under these conditions, the radius of influence will increase with the depth of the sparging point. Radii of influence from 5 feet to 20 feet have been observed in the field by the authors, in coarse materials. In heterogeneous or stratified soils, the authors have observed highly variable radii of influence. In one case, a radius of influence of greater than 60 feet has been observed. The radius of influence is evaluated based on observed increases in soil-gas VOC concentrations in vadose zone monitoring points above the sparging point location, recorded increases in dissolved oxygen levels in saturated zone monitoring points, and localized water-table mounding observed above gas injection points. Under ideal conditions, the achievable radius of influence is limited by the operating pressures that will produce fracturing or short circuiting of the air flow through the formation and/or an excessively turbulent air flow

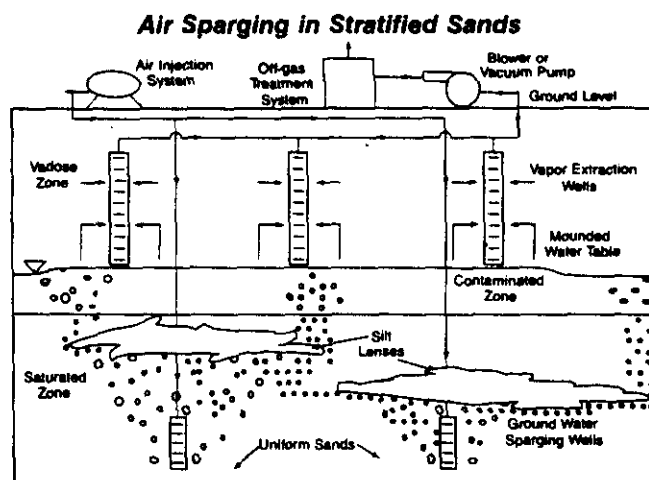


Figure 2. Typical vertical gas channeling in heterogeneous soils.

regime. Further discussion of the achievable radius of influence under specific operating conditions is presented in the case studies that follow.

System Requirements and Limitations

Air-Sparging Equipment

The typical air-sparging system consists of an oil-free air compressor manifolded to one or more air-sparging wells. Expected full-scale operating parameters may be evaluated through the performance of pilot tests using a small compressor at each air-sparging well. This allows for evaluation of the site-specific achievable gas injection flow rates, pressures, radii of influence and vertical channel distribution. A compressor rated for continuous duty at the maximum expected flow rate and pressure will provide the most flexibility in full-scale system operation. Reciprocating compressors and rotary screw compressors are the two most widely used types in this application. Oil-free reciprocating compressors are readily available, but are typically rated for intermittent duty only. Continuous-duty, oil-free reciprocating compressors are available, however they may cost nearly twice as much as intermittent-duty compressors. Rotary screw compressors are typically rated for continuous duty, but are not oil free. Coalescing and particulate filters, and air dryers are available and can clean injection air to less than 3 parts per billion total hydrocarbons. These filtration systems, however, add complexity and maintenance costs. The event of a filtration system failure should also be considered.

Pressurized air is supplied to the sparging wells via a manifold network. Metal pipe or rubber air hose may be used depending on the site-specific conditions. The use of rigid PVC pipe in air-sparging manifold lines should be avoided as the heat generated during air compression can damage the pipe. Where multiple sparging wells are used, a header-type distribution system is used. A pressure gauge and regulator should be provided at each sparging well as a means of measuring and controlling air flow rates and maximizing system flexibility. Pressurized air contained in the formation may force water up the well following system shut-off. Check

valves are used in the manifold line at the wellhead connection to prevent manifold line fouling from this return air flow.

Air-sparging wells may be constructed of rigid PVC or metal casing and screen. The installation of air-sparging wells may be the most costly and difficult aspect of system installation. The presence of running or heaving sands may require the use of drilling fluids to maintain borehole integrity during installation. The well screen or air diffuser must be sealed within a sand filter pack at the design depth. The well's annular seal may be constructed of bentonite pellets or a thick, non-shrinking, neat cement grout. Any cracks or bridging in the seal will allow short circuiting of air flow through the borehole and can greatly reduce the effectiveness of the sparging well.

Potential Advantages of Air Sparging

An air-sparging system, if properly designed, installed, and operated, can potentially provide the most expedient and cost-effective means of saturated zone soil and ground water remediation. By mobilizing aqueous and non-aqueous phase VOCs in the saturated zone and transporting them to the vadose zone, air sparging is essentially an in situ air-stripping system for direct source removal. An advantage of the injection of air as the gaseous medium, into the saturated zone is that it provides a source of oxygen that will be available for hydrocarbon-using bacteria that may be present, thereby stimulating biodegradation of the target contaminants. Where ground water control is not required as part of the full-scale system design, initiation of air sparging does not require the costly and time-consuming procedure of obtaining a water discharge permit.

Potential Disadvantages/Limitations of Air Sparging

In the application of in situ air sparging technology, it is imperative that the overall site remediation plan include a properly engineered soil-vapor extraction system to capture the contaminated vapors emanating from the saturated zone. The potential liabilities associated with the enhanced migration of contaminant vapors off site due to the application of in situ air sparging are obvious. Therefore, the application of air sparging is generally limited to sites where soil-vapor extraction is feasible. One possible exception to the requirement of an accompanying soil-vapor extraction system is a situation in which the overall remediation system design relies on in situ biodegradation to destroy the contaminant vapors in the vadose zone.

The effectiveness of air sparging is sensitive to the lithology and stratigraphy of the saturated and unsaturated zones. In highly stratified soils, air may travel far from the well along coarser strata before reaching the vadose zone, potentially not affecting the target contaminant areas. The lateral migration of the air within the saturated zone will generally be accompanied by a lateral spread in the dissolved contaminant plume. The overall remediation system design should incorporate measures to control the potential contaminant plume spread.

In situations in which DNAPLs are present, air-sparging activities have been observed to spread the immiscible phase and increase the size and concentrations of the VOC plume. This may actually be used as an advantage in a site remediation through the mobilization of the residuals and, in conjunction with ground water control, the realization of a more efficient mass removal process. Although not yet documented, it is likely that air sparging will not be a cost-effective alternative in fine-grained materials such as clays. Additionally, the potential geochemical changes incurred through the application of the technology may cause clogging of the aquifer. The potential for fouling may be evaluated using available geochemical models and avoided by using a more appropriate gaseous medium.

Case Studies of the Application of Air-Sparging Technology

30-Day Air-Sparging Pilot-Scale Evaluation, Connecticut

A 30-day pilot-scale SVE/air-sparging evaluation was performed at a VOC (primarily trichloroethylene) spill site. The pilot SVE/air-sparging system consisted of seven air-sparging wells screened within and below the contaminated soils in the saturated zone at depths up to 15 feet below the static water table, and two vapor extraction wells screened in the unsaturated zone. The air-sparging wells were operated intermittently at air injection flow rates of 3 to 10 scfm and pressures of 15 to 60 psi. Saturated zone soils were generally characterized as stratified fine to very fine sand, with traces of silt. The results of the study showed that air sparging provided a means of enhancing mobilization of the DNAPLS. Approximately 4 pounds of VOC were removed over the 30 days of intermittent operation of the system. However, due to the presence of fine-grained layers, preferential horizontal air flow developed that caused lateral migration of the dissolved-phase VOC plume. Air releases to the vadose zone were observed in monitoring wells as far as 60 feet from the injection wells. Accordingly, the implementation of a full-scale air-sparging system would have required the use of properly engineered hydraulic controls and saturated zone air release wells in order to prevent mobilization and displacement of VOCs off site.

Gasoline Spill Site, Rhode Island

Air sparging was used to expedite ground water remediation at a gasoline spill site in Rhode Island. Ground water/product recovery and soil-vapor extraction had been used at the site from 1985 to 1989 and were successful in removing free product and remediating vadose zone soils to non-detectable (less than 5 parts per billion (ppb)) total benzene, toluene, ethylbenzene, and xylenes (BTEX) levels. However, total BTEX concentrations in ground water remained at approximately 20,000 to 30,000 ppb (benzene at 225 ppb) in the vicinity of the former underground storage tank pit. Figure 3 presents a plan and isopleths for BTEX in ground water at the site. The Rhode Island Department of Environmental Management had set a varying closure limit of

up to 10,000 ppb (at MW-3) total BTEX in ground water for the site. In August 1989, the site was evaluated for the application of air sparging in conjunction with the existing soil-vapor extraction system to expedite site remediation. Based on the parameters evaluated during pilot scale testing, a full-scale air-sparging system was designed and installed at the site consisting of seven shallow sparging wells and six deep sparging wells. Shallow sparging wells were installed in coarse sand and gravel which extended from the water table to approximately 6 feet below the water table. The deep sparging wells were installed in a fine to very fine sand beneath the coarse sand and gravel. The system was operated intermittently over a 60-day period at injection flow rates of 2 to 6 scfm and pressures of 6 to 8 psi. Approximately 5 to 10 pounds of gasoline-range hydrocarbons were stripped from the ground water and saturated zone soils. Within the first two to three weeks of operation, the designated closure criteria had been achieved. Figure 4 presents the changes in total BTEX concentrations in ground water as a result of the application of air sparging at the site. Except for an anomalous rise in total BTEX levels in October 1990, the closure criterion set for the site has been maintained. The levels achieved and maintained at the site are on the order of 600 ppb total BTEX, with non-detectable levels (less than 0.5 ppb) of benzene.

Industrial Facility, Connecticut

In January 1991, feasibility testing for SVE/air sparging was conducted on a site in south-central Connecticut. The objectives of the field testing were to evaluate the feasibility of removing saturated zone soil contamination trapped below the water table using air sparging in conjunction with SVE and, if appropriate, to determine the full-scale conceptual design criteria for the site.

Background information about the site indicated that an undetermined volume of gasoline was released from an underground storage tank in 1983. Initial free-product recovery efforts resulted in the recovery of 600 gallons of product. Hydrogeologic investigations indicated the site was underlain by shallow fill deposits and by deeper glacial till deposits consisting of fine to medium sand with varying percentages of fine gravel, cobbles, and silt. Two separate areas on-site (Areas 1 and 2) located approximately 140 feet apart were identified as containing free-phase product or elevated concentrations of dissolved VOCs. Depths to the water table at the site ranged from 5 to 13 feet below ground surface. The large variations in water table depth reflect several abrupt changes in surface elevation.

Based on available historical data, an SVE/air-sparging pilot test design was developed, which included the drilling and installation of three vapor extraction wells, two air-sparging wells and 10 soil-vapor monitoring probes. Headspace screening conducted during drilling operations with a photoionization detector (PID) indicated that saturated zone soil contamination ranging in concentration from 180 ppm-v/v (parts per million on a volume per volume basis) to 300 ppm-v/v extended from the water table to a maximum depth of 20 feet

Total BTEX Concentration Isopleths in Ground Water (May 1987)

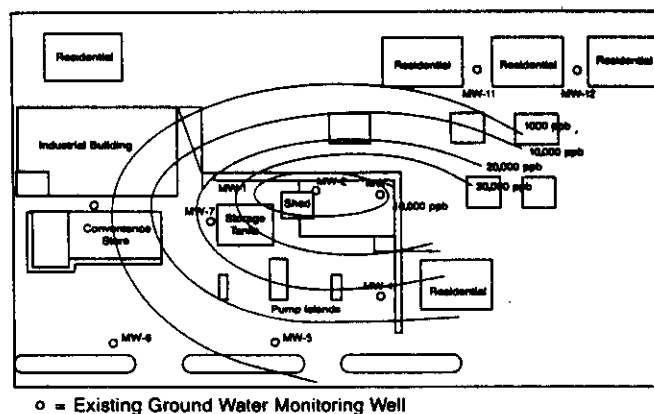


Figure 3. Site plan and BTEX isopleths for air-sparging site in Rhode Island.

TOTAL BTEX CONCENTRATION AT MW-3 SAMPLING PRIOR TO/AFTER AIR INJECTION

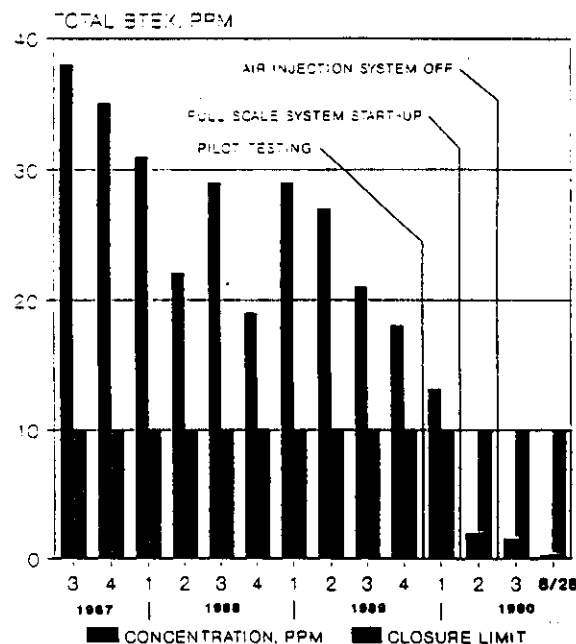


Figure 4 Changes in total BTEX levels at Rhode Island site as a result of the application of air-sparging technology.

below ground surface. The air-sparging wells were installed using 1.5-inch diameter PVC screen and riser, and a 1-foot long screened interval located between 17 and 19 feet below ground surface. SVE discharge concentrations were measured with a portable gas chromatograph, which was equipped with a flame ionization detector (GC/FID).

SVE/air-sparging testing in Area 1 was conducted for approximately eight hours at an SVE flow rate of 24 cfm. Initial SVE discharge sample analysis (via GC/FID) in Area 1 showed a gradual decrease in total VOC concentrations from 5471 ppm-v/v to 3996 ppm-v/v prior to air sparging. Upon application of air sparging in Area 1, SVE discharge sample analysis resulted in a

sharp increase in total VOC concentrations to 11,393 ppm-v/v followed by a gradual decline to 9047 ppm-v/v over a four-hour period. Figure 5 presents the measured VOC concentrations in the SVE system discharge during the testing of Area 1. Air injection rates and pressures at each sparging well ranged from 1 to 3 scfm at an average of 17 psi during testing in Area 1.

The results of SVE/air sparging field testing at the site indicated that there is a significant volume of VOCs trapped below the water table as residual soil contamination and that SVE/air sparging is an appropriate means of removing these compounds. Field testing also resulted in the design of a full-scale system consisting of 12 air-sparging points and three SVE wells in each area, with sparging points installed on 12- to 15-foot centers.

Retail Gasoline Station, Massachusetts

From the period March 1989 through May 1990, an SVE system was installed and operated at a retail gasoline station in southeastern Massachusetts. The objective of the SVE remediation system was to recover gasoline-range VOCs existing as free product and residuals from vadose zone soils at the site. During the period of operation, the SVE system successfully recovered more than 600 gallons of gasoline from the site.

Based on ground water quality sampling and analytical data, it was determined that significant residual soil contamination was present below the water table. Under a limited budget of \$30,000, from the period October 1990 to January 1991, an SVE/air-sparging system was installed and operated at the site. Nine 1-inch diameter PVC air-sparging wells were installed with standard auger drilling techniques. The wells were installed with 1-foot screened intervals at depths from 15 to 19 feet below ground surface. The water table depth fluctuated seasonally between 8 and 10 feet below ground surface. Site soils consisted of clean, well-sorted, fine to medium sands to a depth of greater than 20 feet. The SVE/air-sparging equipment consisted of a 1.0 horsepower regenerative blower (98 cfm maximum capacity), a timer-controlled 2.5-horsepower oil-less reciprocating compressor, and a catalytic oxidation unit (100 cfm maximum capacity). Air-sparging manifold lines consisted of flexible 3/8-inch high-pressure hose, stainless steel ball valves and pressure regulators. Figure 6 presents a site plan showing the SVE/air-sparging layout.

The air-sparging system was configured to simultaneously operate a group of three air injection wells for two four-hour cycles over a 24-hour period. This operating configuration was chosen to minimize the possibility of preferential air flow channeling resulting from the air-sparging process while maximizing hydrocarbon removal and energy consumption efficiency. The air-sparging well groups were alternated frequently to maintain maximum VOC removal rates.

Pressures ranging from 4 to 6 psi at flow rates of 3 to 5 scfm were measured during operation of the air-sparging system. Ground water monitoring results indicated that the water table was not significantly affected

Pilot Test for Air Sparging - Area 1 Industrial Facility, CT

SVES Discharge Concentrations (ppm-v/v)
(Thousands)

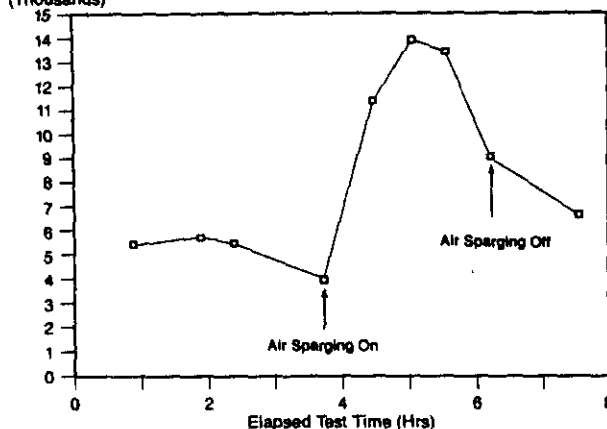


Figure 5. Soil-vapor extraction system discharge from Area 1 prior to and during air-sparging pilot testing at an industrial facility.

Retail Gasoline Station - Massachusetts Site Plan

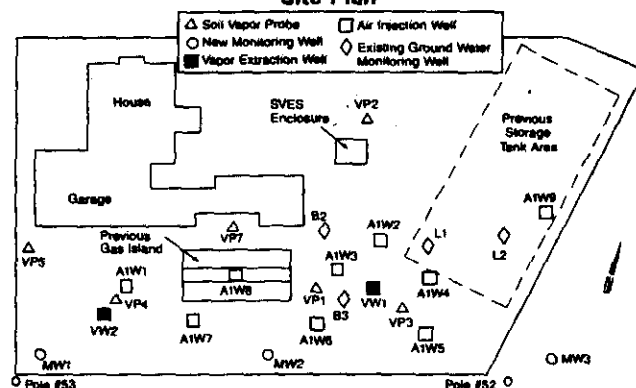


Figure 6. Site plan and SVES/air-sparging system layout retail gasoline station in Massachusetts.

Retail Gasoline Station - Massachusetts

Full Scale SVES/Air Sparging

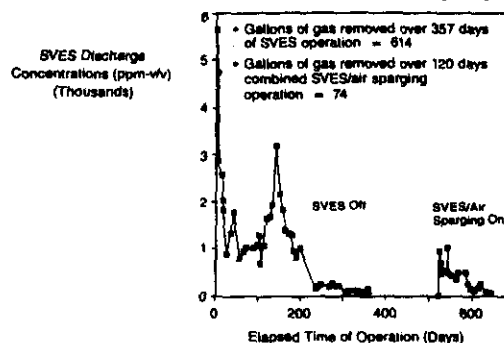


Figure 7. Graph showing SVES system discharge VOC concentrations prior to and during air-sparging activities at the site.

by the operation of the SVE/air-sparging system. During the air-sparging operation, initial SVE system discharge concentrations were 931 ppm-v/v and the final SVE system discharge concentrations were 65 ppm-v/v. A baseline concentration of 100 to 65 ppm-v/v from the sole operation of the SVE system was recorded during the operating period.

It is estimated that the SVE/air-sparging system removed approximately 67 to 74 gallons of gasoline during the test period, of which 7 to 12 gallons were directly attributable to the air-sparging system. Following shut-down and demobilization of the SVE/air-sparging system in January 1991, ground water quality samples were collected and analyzed in January and April 1991, from monitoring wells for which historical water quality data existed. Laboratory analytical results indicated that concentrations of total BTEX concentrations in ground water declined from approximately 50,000 ppm-v/v to less than 10,000 ppm-v/v and are continuing to decline. Ground water sampling fewer than 100 feet downgradient from the site did not detect BTEX, suggesting that no contaminant spread laterally as a result of the air-sparging system operation. Based on the estimated removal rates achieved with the SVE/air-sparging system and the declines in dissolved concentrations of VOCs in ground water, it appears that the objective of removing the major saturated-zone source areas have been met within the limited financial resources available for the project.

Summary and Conclusions

The combined use of SVE and air sparging has been demonstrated to be a successful and cost-effective integrated remediation technology for removing VOC sources from unsaturated and saturated zone soils. The technique is applicable to either gasoline-range VOCs or DNAPLS and is effective for removing product adsorbed to soils, free-phase product and dissolved VOCs in ground water. Benzene levels in ground water have been reduced through the sparging process to less than 1 ppb. The current state-of-the-art design process for air sparging is largely empirical due to the numerous variables encountered and the complex multifluid flow processes occurring. Studies are currently being performed on bench-scale and field-pilot scale to better establish engineering design criteria for full-scale air-sparging systems.

The advantages of air-sparging technology are that it can provide expedient and cost-effective in situ remediation of saturated zone VOC contamination. In some cases, dewatering is not necessary to expose contaminants in saturated zone soils, therefore, pump-and-treat systems and delays for obtaining water discharge permits are not incurred. If ground water control is required at a site, the combined application with air sparging can enhance pump-and-treat recovery of saturated zone contaminants because air sparging can aid in the dissolution/mobilization of immiscible phase VOCs. Air sparging can also be used to provide an oxygen source to a typically anoxic environment, which, in turn, may stimulate naturally occurring biological degradation of the contaminants.

The potential limitations of air sparging are that it will generally be effective only with VOCs, which are amenable to air stripping (excepting those remediated through the enhanced biodegradation process) and with soils that are suitable to effective SVE. Additionally, air sparging generally should be used in conjunction with

SVE to capture potential fugitive vapors from the air-sparging process. The misapplication of the air-sparging process also carries the risk of displacing dissolved VOCs in both vertical and horizontal directions, which may spread a contaminant plume beyond the pre-sparging limits. In these cases, ground water pumping and/or injection wells can be used to manage plume migration.

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Biographical Sketches

Michael Marley is the vice president of technology and a principal at VAPEX (Vapex Environmental Technologies Inc., 480 Neponset St., Canton, MA 02021). He has a bachelor's degree from Queens University (Belfast, Northern Ireland) and a master's degree from the University of Connecticut, both in civil engineering. Marley is completing work on his Ph.D. dissertation at the University of Connecticut on the development and application of air flow models in the design of soil and ground water remediation systems. He has more than 13 years of related experience, has lectured nationally on the design and implementation of soil-vapor extraction systems and complimentary technologies, and has authored numerous technical papers.

David Hazebrouck is currently a project manager at VAPEX (Vapex Environmental Technologies Inc., 480 Neponset St. Canton, MA 02021). He has a bachelor's degree in geology from the University of Rhode Island. Hazebrouck has more than seven years experi-

ence in hydrogeological studies, hazardous waste site assessments, and the design and implementation of ground water and soils remediation systems. With VAPEX, he is responsible for project management activities associated with soil-vapor extraction and air-sparging field pilot tests; regulatory negotiations; and full-scale soil-vapor extraction and air-sparging systems design, installation, and operation.

Matthew T. Walsh is a project geologist/manager at VAPEX (Vapex Environmental Technologies Inc., 480 Neponset St., Canton, MA 02021). He has a bachelor's degree from LaSalle University (Philadelphia, Pennsylvania) in geology and a master's degree from the University of Lowell (Massachusetts) in environmental studies. Walsh's thesis study at Lowell was titled "Mobilization of Various Organic Contaminants in Fine Grained Sediments by Electro-Osmosis." At VAPEX, he is responsible for numerous site investigations and the field pilot testing, design, installation, and monitoring of several full-scale vapor extraction and air-sparging systems.

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Air Sparging in Gate Wells in Cutoff Walls and Trenches for Control of Plumes of Volatile Organic Compounds (VOCs)

by James F. Pankow^a, Richard L. Johnson^a, and John A. Cherry^b

Abstract

Volatile organic compounds (VOCs) can be stripped from ground water by sparging air into water in wells or in trenches. This well/trench sparging ("WTS") can remove VOCs from plumes of contaminated ground water as that water passes across the sparge zone. With sparging in wells, cutoff walls will be needed to force the contaminated water through the "gate" wells. With in situ sparging ("ISS"), air is sparged directly into a contaminated aquifer. ISS may be useful in treating local zones of high contamination, but WTS is better suited for treating large plumes of contaminated ground water. Interest in sparging methods is growing because: (1) they do not remove water from the subsurface, and so difficult disposal issues are avoided and an increasingly valuable water resource is not depleted; and (2) the Darcy velocity v in many systems is low, and so only a relatively small volume of water must be treated per unit time.

The theoretical fractional efficiency of WTS is given by $E = S/(1 + S)$. The parameter S is named here as the "dimensionless sparge number," with $S = HR_g/(RTyv)$ where H (atm-m³/mol) = Henry's Law constant for the compound of interest; R_g (m³/s, at 1 atm pressure) = gas sparging rate; R = gas constant ($= 8.2 \times 10^{-5}$ m³-atm/mol-deg); T = temperature (K); yz (m²) = cross-sectional area producing the water which is passing into the sparging zone; and v = Darcy velocity (m/s). E increases as S increases. E increases as H increases because the volatility of a compound goes up as its H increases. Plots for E in WTS are given vs. S as well as vs. some of the variables making up S .

Well/trench sparging (WTS) has the potential to become a useful treatment method for removing VOCs from contaminated ground-water plumes. It is suited for use with most of the solvents and petroleum products which have caused extensive ground-water contamination. The theory of the method is simple, and the theoretical removal efficiencies are predictable as well as adjustable.

Introduction

General

The contamination of ground water by volatile organic compounds (VOCs) remains an important environmental problem. VOCs of interest include many dense nonaqueous phase liquids ("DNAPLS," e.g., the chlorinated solvents), as well as the petroleum-related "BTEX" group (benzene, toluene, ethylbenzene, and the xylenes). Most current efforts to remediate or at least control VOC contamination involve pumping at purge wells. Accelerated in situ biodegradation has proven difficult on large scales, accelerated dissolution using micelle-forming surfactants is still under development, and aquifer excavation is usually far from

practical. With pumping at wells, the water may be: (1) discharged directly to a sewer; (2) treated and discharged to a sewer; or (3) treated and returned to the aquifer. With sewer discharge, an often only slightly contaminated natural resource is discarded. With aboveground treatment with subsequent return to the aquifer, there may be significant capital costs. Therefore, alternative treatment technologies remain of interest. This is especially the case for *plume control*, given that full remediation is often essentially impossible at many sites of interest.

It is natural to try and take advantage of the volatility of VOCs by designing aeration methods which remove them directly from the subsurface. Certainly vacuum extraction has been found useful in removing VOCs from the vadose zone. For the saturated zone, aeration can take place by two methods. In "sparging" in a well or a trench (abbreviated here as "WTS"), air is injected into *liquid* water at the bottom of an open well or trench (Figures 1a, 2a, 2b). The freely rising bubbles of air strip volatile compounds from the ground water that is flowing through the sparge zone. Depending on the situation, the VOCs in the gas generated by sparging may need treatment prior to release to the atmosphere. In an early application, Coyle et al. (1985) used

^aDepartment of Environmental Science and Engineering, Oregon Graduate Institute, 19600 N.W. Von Neumann Dr., Beaverton, Oregon 97006.

^bWaterloo Centre for Groundwater Research, Department of Earth Sciences, University of Waterloo, Waterloo, Ontario, Canada N2L 3G1.

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In contrast to WTS, in "*in situ sparging*" (ISS), air is injected directly into a saturated *porous medium* through an installed screen (Marley et al., 1992; Brown and Jasiulewicz, 1992) (see Figure 1b). VOCs are thereby removed directly from the aquifer. The gas flow generated by ISS can be coupled to a vacuum extraction operation involving the vadose zone. In sandy aquifers, at low ISS air injection rates, stable channels of air will be established in the medium; the air rising through the channels will remove VOCs from water in the adjacent pores, as well as from any volatile liquid product phase that might also be present in those pores. In coarse gravels, the injected air may rise as bubbles towards the water table. At high air injection rates in sandy, shallow, water-table aquifers, the possibility exists that the saturated zone near the injection point could become fluidized, with bubbles then rising towards the surface. Fluidization of a *portion* of the aquifer in a zone where a liquid organic contaminant is present could also cause a mobilization of that organic liquid. This can be helpful if the liquid is volatile because it will facilitate the vaporization process. However, ISS might also thereby lead to an increased rate of dissolution and transport away from the treatment zone, including possibly introducing small droplets of the liquid organic phase into the moving ground water.

The use of sparging in wells to remediate a site contaminated with VOCs has been suggested by Herrling et al. (1990), Herrling and Buermann (1990), and Gvirtzman and Gorelick (1992). The air lift induced by the sparging is used to pump water upwards for immediate reinfiltration. In the approach of Herrling et al. (1990) and Herrling and Buermann (1990), the reinfiltration occurs through a screen near the top of the well. In the approach of Gvirtzman and Gorelick (1992), the reinfiltration occurs through a circular gallery installed near ground surface. In both approaches, the reinfiltration creates a continuous circulation between the well and the aquifer. With each passage through the sparge well, the levels of the VOCs are reduced. As with other sparging methods, no net water is removed from the aquifer. Presumably, numerous of these recirculating sparge wells would be needed to remediate any real site. A disadvantage of this approach is that as the depth to the water table increases, the need to bring water close to the surface for distribution in the infiltration gallery will require sparge rates that are increasingly large. When examined on a mass of VOC stripped per unit volume of air basis, these rates may become unattractive.

The diagram illustrates a remediation system for a contaminated aquifer. At the top, the 'atmosphere' is shown with a sun and a cloud. On the ground surface, an 'air compressor' is connected to a well. Air is pumped down the well, creating a series of upward-pointing arrows that represent air rising through the aquifer. These arrows are labeled as 'zone containing channels of air moving upward through the sat'd. zone'. The well extends to the 'bottom of aquifer, or lower boundary of contamination'. The 'ground surface' and 'water table' are also indicated.

Fig. 1b. Direct air injection using in situ sparging ("ISS") in the saturated zone of an aquifer in a porous medium (cross-sectional view). Air moves upwards through channels (represented as lines) in the porous medium.

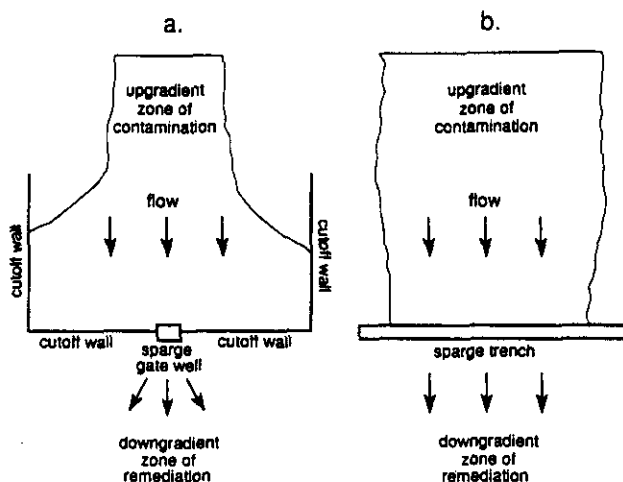


Fig. 2. a. Sparging in a gate well in a cutoff wall for interception and treatment of a flow of contaminated ground water (plan view). Note the tendency of the plume to widen behind the cutoff wall. b. Sparging in a rectangular trench for interception and treatment of a flow of contaminated ground water (plan view).

dispersed zones of VOC contamination, removal by ISS may be difficult because of inefficiencies in the transport of the VOCs to the injection zone. This may remain true even when those areas of contamination lie directly upgradient of the ISS area. Indeed, note that irregularities in the distribution of the air channels (at low ISS flow rates) or in the zones of fluidization (at high ISS flow rates in shallow sandy aquifers) may allow advected water to flow through the general area of the injection zone and yet miss exposure to the air channels. For sites at which a liquid DNAPL solvent is present in one or more pools directly on top of an aquitard, treatment of the ground water by ISS is not likely to be effective. Indeed, as shown by Johnson and Pankow (1992) and Anderson et al. (1992), dissolution from a DNAPL pool into the overlying ground water is normally very slow. In addition, it will be very difficult to get much direct ISS air contact with a DNAPL pool.

Control and Remediation of a VOC Plume

When an entire VOC contamination site cannot be remediated in a cost-effective manner by any means, then long-term plume control is often the only remaining option. Preferably, this option will involve remediation of the ground water that does leave the area. ISS will probably not be useful in this mode because of irregularities in the air injection process. Sparging in a continuous trench in WTS, however, could be used to remove VOCs from a moving plume (Figure 2b). Under most natural ground-water flow conditions, the volume flux of water through an aquifer (given by the Darcy velocity) is quite small. In the types of unconfined sand or gravel aquifers that are prone to widespread contamination, typical volume fluxes are 0.015 to 0.50 m³/m²-day in the direction of flow (Darcy velocities of 0.015 to 0.5 m/day). This range corresponds to 0.010 to 0.35 liters/m²-min. Thus, even for aquifers that are tens of meters thick, the volume flow rate per meter of aquifer width is less than a few liters per minute. This is a relatively small rate in a treatment context.

A trench for WTS could be constructed using perforated, interlocking sheet-piling. After installing the two sheet-pile walls, the zone between the walls would be excavated and the gas lines for sparging installed. Since the concentration of the contaminant in the ground water arriving at the trench will surely vary along the length of the trench, it may also be advantageous to subdivide the cells along that length. The flow of sparge air could then be adjusted along the length of the system so that the majority of the air is applied where it is needed the most.

An alternative to an open sparge trench would be one that is backfilled with coarse gravel (or pebbles). Gravel-filled trenches can be installed using conventional technology. In the case of cohesive materials, direct excavation could be carried out. In the case of noncohesive materials, driven sheet-piling could be used. Following excavation, the air lines could be laid, and the zone backfilled with the desired material. This type of system would be easier to stabilize against caving than would an open sparge trench.

Sparging in wells could also be used to treat a plume, but in this mode, cutoff walls will surely be needed to force all of the contaminated flow into the sparge "gate well" (Figure 2a). The types of cutoff walls which could be used include conventional bentonite slurry walls installed by trenching, or the type of sealable, driven sheet-piling that has been described by Starr et al. (1992). Our field research indicates that gate wells can be constructed from a rectangle of perforated, interlocking sheet-piling that is, in turn, locked into the cutoff wall. Slotted baffles could divide the sparge zone into sequential cells (see below), with sparge heads placed at the bottom of each cell. If the cutoff wall is of interlocking, sealable sheet-piling, then the sparge zone could be locked directly into the cutoff wall. If the cutoff wall is of bentonite, the sheet-piling could be withdrawn after the sparge cell is installed, and the sparge gate sealed to the cutoff wall with additional bentonite.

Because of the tendency of the plume to build up and spread behind a cutoff wall, the plume width which a given system will be able to handle will be less when a cutoff wall/sparge gate(s) system is used than when a sparge wall is used. However, for a wide contaminant plume, a cutoff wall/sparge gate combination might be installed at less cost than a sparge trench. In order to prevent an excessive rise in the hydraulic head upgradient of the gated cutoff wall, results from our modeling work suggest that in the Figure 2a configuration, to avoid too much hydraulic head buildup, it will be necessary to have 5-20% of the wall as gates. The focus of this paper will be on the use of WTS for the control and remediation of VOC plumes.

Historical Perspective on Sparging

Up until the mid 1980s, the development of sparging for use in either remediation or in plume control was inhibited by the view that it was easier to treat VOC-contaminated water after it was removed from the subsurface. For example, the flow of the VOC-contaminated water can be controlled more easily in pump and treat, and the energy requirements of pump and treat with cascade aeration are generally lower per volume of water treated than with sparg-

ing. With respect to the latter point, note that: (1) pumping a given volume V of air a distance h meters below the water table so as to create bubbles for sparging will take roughly the same amount of energy as pumping the same volume V of water a similar distance above the water table for cascade aeration; and (2) it is relatively inexpensive to blow large volumes of air against the low backpressures in an aeration tower, and so in cascade aeration, a given volume V of water can be exposed to a much larger volume of air for stripping the VOCs than can be accomplished using a volume V of air during well sparging. Therefore, from a simple point of view that considers only the cost-effectiveness of the treatment, assuming roughly similar capital equipment needs, pumping followed by cascade aeration will generally be more economical than well sparging.

Much has changed, of course, since the early 1980s. Now and for the foreseeable future, the simple economics of treatment is by far not the only consideration involved in deciding on treatment options. Issues of *public perception*, *regulatory policy on the disposal of contaminated water*, as well as a greatly increased *water resource value* have converged to make sparging much more attractive for use in plume control. For example, consider a ground-water plume that contains a mean concentration of trichloroethylene of $100 \mu\text{g/l}$ (100 ppb). Let us say that a certain WTS sparging design is capable of reducing the mean concentration to the current U.S. EPA drinking water limit of $5 \mu\text{g/l}$. In many communities (especially those in the drier climates of the west), it might well be easier to win approval for this in situ, 95% efficient treatment approach than it would be to gain approval for a design which pumps the water from the ground, treats it with 95% efficiency, then: (a) injects or infiltrates the water, viewed as "still-contaminated" back into the aquifer; or (b) discards the resource into a surface stream or a sewer. Thus, the energy disadvantage of sparging can quickly become a nonissue. Furthermore, we note that pump and treat methods often draw uncontaminated water into the zone of contamination, and also that abilities of such methods to remove contaminants from an aquifer generally decrease in time.

Plume Control and Remediation by Well/Trench Sparging (WTS)—Theory

Figure 3 is a schematic diagram of sparging taking place in a single gate well, or along a single trench across the zone of contamination. Due to difficulties in construction, a depth of ~ 50 m probably represents an upper limit for the installation of WTS trench zones.

Sparging with air will lead to a saturation of the ground water with oxygen. When the water of interest is anoxic, this may lead to the problematic precipitation of iron and manganese oxyhydroxides in, as well as downgradient of, the sparge zone. Since contaminated ground water is frequently close to the ground surface, for the many systems of interest that are already largely oxic, there will be no such precipitation. Moreover, since biological degradation of some compounds is promoted by oxygen, raising the oxygen levels to saturation can be beneficial in further lowering the contamination that escapes removal by sparging. If oxygenation of

an aquifer needs to be avoided, then sparging in a closed system can be carried out as described by Herrling et al. (1990). In that approach, the sparge air is recycled. Contaminants are cleaned from the air using activated carbon, and the cleaned air is reused. The anoxic water in the well quickly removes the oxygen from the air, and the sparging then continues using the nitrogen and other inert gases remaining in the air.

The theoretical efficiency of WTS in removing *dissolved* volatile contaminants from the water which actually passes through a sparge zone may be predicted using a constant flow, stirred tank reactor (CFSTR) approach (e.g., see Levenspiel, 1972). The predictions require a knowledge of the Henry's Law constant H ($\text{atm}\cdot\text{m}^3/\text{mol}$) for the compound of interest, the gas sparging rate R_g (m^3/s , at 1 atm pressure), the cross-sectional area yz (m^2) producing the water which is passing into the sparging zone, and the Darcy velocity v (m/s).

Zone A in Figure 3 represents the portion of an aquifer that is contaminated at the dissolved concentration c_i (mols/m^3) with a VOC. When Zone A is characterized by a range of concentrations, the volume-averaged mean concentration *entering* Zone B should be used as c_i . Zone B is the well-mixed, open well (or trench) in which single stage sparging is occurring. (The manner in which the removal efficiency can be increased by distributing R_g over several *sequential* stages is discussed below.) The concentration in Zone B is c_f (mols/m^3). Zone C is the zone of the aquifer that is receiving the treated (c_f) water. The use of a single c_f value to describe the concentration in the sparging zone results from the assumption that the sparging zone is well-mixed. Bubble zones in liquid water columns are indeed well-mixed vertically, and the level of in-column dispersion increases with the sparge gas flow rate (Siemes and Weiss, 1959). The

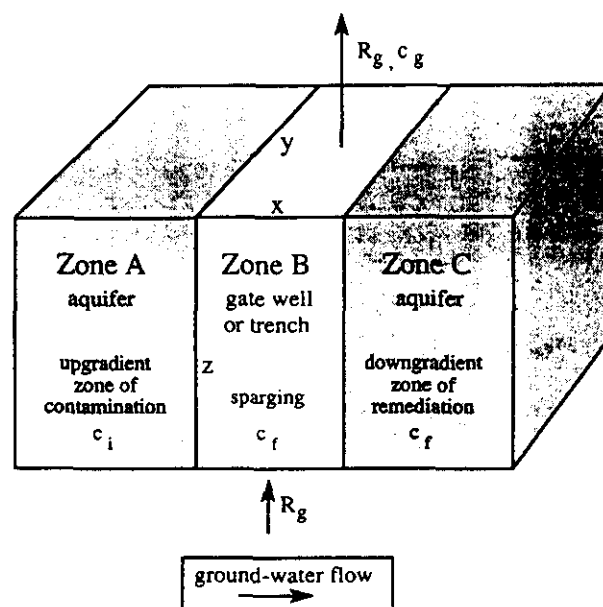


Fig. 3. Model for contaminant removal by sparging with a single stage. Contamination at concentration c_i in the upgradient portion of the aquifer (Zone A) is lowered in the sparging zone (Zone B) to c_f before flowing into the downgradient portion of the aquifer (Zone C) at concentration c_f .

bubbles in Zone B will pose essentially no resistance to flow through the zone.

At steady state, c_f is constant in time. The ground-water volume flux ($\text{m}^3/\text{m}^2\text{-s}$) is given by the Darcy velocity v (m/s). In units of mols/s , the mass balance for a given VOC on the sparging zone is then

$$\begin{matrix} \text{in} & \text{out} \\ c_i y z v = c_f y z v + c_g R_g \end{matrix} \quad (1)$$

where $y z v$ is the water volume flow rate into and out of the sparging zone, and c_g is the concentration (mols/m^3) of the VOC in the sparge gas leaving the sparging zone. In the case of a gate well in cutoff wall, y is the effective capture width of the gate well. According to the Ideal Gas Law,

$$c_g = n/V = p/RT \quad (2)$$

where n/V = number of mols of volatilized contaminant per m^3 of sparge gas, p is the partial pressure of the contaminant in the sparge gas (atm), R is the gas constant ($= 8.2 \times 10^{-5} \text{ m}^3\text{-atm}/\text{mol-deg}$), and T is the temperature (K). Note that if t is the temperature in degrees centigrade, then $T = t + 273.15$.

In WTS, it may be expected that equilibrium will be nearly established in the sparge zone between the ground water and the sparge gas. The equilibrium relationship between the partial pressure of a gas and its aqueous concentration is given by Henry's Law as

$$p = H c_f \quad (3)$$

The larger the value of H , the greater the volatility of the compound. Using an approach discussed by Gvirtzman and Gorelick (1992), it can be shown that except for high sparge rates, the equilibrium for VOCs represented by equation (3) will in fact be attained. Fair et al. (1973) also discuss this matter in considerable detail.

By equations (1)-(3),

$$c_i y z v = c_f y z v + (H c_f / RT) R_g \quad (4)$$

$$c_f / c_i = 1 / [1 + H R_g / (R T y z v)] \quad (5)$$

We define S as a dimensionless "sparge number":

$$S = H R_g / (R T y z v) \quad (6)$$

so that in the ideal case,

$$c_f / c_i = 1 / (1 + S) \quad (7)$$

The theoretical ideal fractional efficiency E of the WTS process is given by

$$E = 1 - c_f / c_i \quad (8)$$

$$= S / (1 + S) \quad (9)$$

The larger S is, the more efficient is the removal. Thus, E increases with increasing H since the volatility of a compound goes up as its H value increases. Increasing R_g will also increase E . Increasing y , z , and v , however, both decrease E since they increase the volume rate of addition of contaminated water to the sparging zone. The dependence of E on R_g , y , z , and v is contained in the dependence of E on $R_g / y z v$, the dimensionless air to water ratio. Since a given

zone or subzone of contamination under treatment will be characterized by specific values of y , z , and v , then for that zone or subzone of the parameters making up $R_g / y z v$, only R_g may be adjusted to improve E .

We note that in the ideal, theoretical case, the volume of the zone xyz in which WTS is occurring does not affect E ; changing x does not alter the ratio of the volume rate of addition of sparge gas to the volume rate of addition of contaminated water. For example, while doubling x doubles the hydraulic residence time in the sparge zone, and so one might expect an increase in E , the volume of water which R_g must treat is also doubled, and so E remains unchanged. The fact that E is independent of x indicates that x can be varied so as to avoid difficulties associated with trying to force too much air through too little water.

Table 1 summarizes H data for the 20-25°C range for a variety of compounds of interest. In the case of the contamination of an aquifer with more than one compound, to the extent that the different compounds possess different H values, they will be removed with different efficiencies. In addition to being compound-dependent, H values are also temperature-dependent, often increasing by a factor of ~ 2 for every 10 degree increase in temperature. Use of the Table 1 data at temperatures other than 20-25°C should be made cautiously. Therefore, if the Table 1 data are used when the ground water is cooler than 20-25°C, the calculations will produce best-case E values. As noted by Collins (1925), the mean temperature of ground water in the United States ranges between 5°C and 25°C. If $H(288.15 \text{ K})$ is the value of H at $T = 288.15 \text{ K}$ (15°C), then at this temperature

$$S = H(288.15 \text{ K}) R_g / (0.024 y z v) \quad (10)$$

An increasing amount of information is becoming available giving the T -dependence of H values for VOCs of interest (e.g., Hunter-Smith et al., 1983; Gossett, 1986). Whenever possible, H values for the exact temperature of interest should be employed. Sometimes, literature values for H are expressed as dimensionless air to water concentration ratios (i.e., in our notation, they are sometimes expressed as H/RT values); to convert to the units used here ($\text{atm-m}^3/\text{mol}$), multiply by RT ($= 0.024 \text{ atm-m}^3/\text{mol}$ @ 288.15 K).

The criterion for inclusion in Table 1 was a minimum H value of $8 \times 10^{-5} \text{ atm-m}^3/\text{mol}$. Approximately half of the organic EPA "priority pollutants" are in Table 1. Most of the data are reliable. However, as with all equilibrium constants, the accuracies of calculations made based on such data depend upon the reliabilities of the constants themselves. Note that the H values reported for the PCBs are *mixture-average* values. As such, they are not true thermodynamic constants, and should only be used as *general indicators* of the behavior of these PCB mixtures.

All of the petroleum-related monocyclic aromatic compounds are relatively volatile. This fact is very positive from the viewpoint of sparging gasoline-contaminated ground waters. Although not included in Table 1, it may be noted in the same regard that aliphatic compounds (found at very high concentration in gasoline) possess very large H values (Mackay and Shiu, 1981), and moreover are only sparingly soluble in water in the first place.

Table 1. Henry's Law Constants (H , atm-m³/mol) for Selected Organic Compounds
[Data Obtained from Mabey et al. (1982) and Mackay and Shiu (1981)]

Compound	H	t (°C ^a)	Compound	H	t (°C ^a)
Chlorinated Nonaromatics			Monocyclic Aromatics, continued		
Methyl chloride	0.04	20	Hexachlorobenzene	0.00068	20/25
Methyl bromide	0.20	20	Toluene	0.0067	20
Methylene chloride	0.0020	20/25	Ethylbenzene	0.0066	20
Chloroform	0.0029	20	<i>o</i> -Xylene	0.0050	25
Bromodichloromethane	0.0024	20/22	<i>m</i> -Xylene	0.0070	25
Dibromochloromethane	0.00099	20/22	<i>p</i> -Xylene	0.0071	25
Bromoform	0.00056	20	1,2,3-Trimethylbenzene	0.0032	25
Dichlorodifluoromethane	3.0	25	1,2,4-Trimethylbenzene	0.0059	25
Trichlorofluoromethane	0.11	20	1,3,5-Trimethylbenzene	0.0060	25
Carbon tetrachloride	0.023	20	Propylbenzene	0.0070	25
Chloroethane	0.15	20	Isopropylbenzene	0.0013	25
1,1-Dichloroethane	0.0043	20	1-Ethyl-2-methylbenzene	0.0043	25
1,2-Dichloroethane	0.00091	20	1-Ethyl-4-methylbenzene	0.0050	25
1,1,1-Trichloroethane	0.03	25	<i>n</i> -Butylbenzene	0.013	25
1,1,2-Trichloroethane	0.00074	20	Isobutylbenzene	0.033	25
1,1,2,2-Tetrachloroethane	0.00038	20	sec-Butylbenzene	0.014	25
Hexachloroethane	0.0025	20/22	tert-Butylbenzene	0.012	25
Vinyl chloride	0.081	25	1,2,4,5-Tetramethylbenzene	0.025	25
1,1-Dichloroethene	0.19	25/20	1-Isopropyl-4-methylbenzene	0.0080	25
1,2-trans-Dichloroethene	0.067	20	<i>n</i> -Pentylbenzene	0.0060	25
Trichloroethene	0.0091	20			
Tetrachloroethene	0.0153	20	Pesticide and Related Compounds, and PCBs		
1,2-Dichloropropane	0.0023	20	Ethylene dibromide (EDB) ^b	0.00082	25
trans-1,3-Dichloropropene	0.0013	20/25	trans-Chlordane	0.000094	25
Hexachlorocyclopentadiene	0.016	25	Heptachlor	0.0040	25
Hexachlorobutadiene	0.026	20	Heptachlor epoxide	0.00039	25
			2,3,7,8-TCDD	0.0021	—
Chlorinated Ethers			Aroclor 1016 ^c	0.00033	25
Bis(chloromethyl)ether	0.00021	20/25	Aroclor 1221 ^c	0.00017	25
Bis(2-chloroisopropyl)ether	0.00011	20	Aroclor 1242 ^c	0.0020	25
4-Chlorophenylphenylether	0.00022	25	Aroclor 1248 ^c	0.0036	25
4-Bromophenylphenylether	0.00010	20/25	Aroclor 1254 ^c	0.0026	—
Monocyclic Aromatics			Polycyclic Aromatics		
Benzene	0.0055	25	Naphthalene	0.00046	25
Chlorobenzene	0.0036	20/25	Acenaphthene	0.000091	25
<i>o</i> -Dichlorobenzene	0.0019	20	Acenaphthylene	0.0015	20/25
<i>m</i> -Dichlorobenzene	0.0036	25	Anthracene	0.000086	25
<i>p</i> -Dichlorobenzene	0.0031	25	Phenanthrene	0.00023	25
1,2,4-Trichlorobenzene	0.0023	25			

^aWhere two temperatures are given, the first is the temperature at which the vapor pressure was measured, and the second is the temperature at which the solubility was measured.

^bVapor pressure data from Stull (1947), and solubility data from Stephen and Stephen (1963).

^cMixture-average value.

When the Table 1 pesticides and related compounds, PCB formulations, and polycyclic aromatic compounds are present in soils containing nontrivial amounts of organic carbon, they will not be expected to move very rapidly (Karickhoff, 1984). However, in porous media of very low organic carbon (e.g., sands and gravels), and when soil particles and/or emulsions of liquid are facilitating the sub-surface transport of such compounds, their degrees of retardation will be smaller, and plume control by WTS sparging may be of interest.

While all of the compounds in Table 1 are at least somewhat volatile from water, some are substantially more volatile than others. E.g., dichlorodifluoromethane is 35,000 times more volatile from water than is anthracene. The dependence of E upon S is illustrated in Figures 4 and 5.

Efficient removal by sparging is possible for low H compounds for a given v value by adjusting S through the manipulation of R_g/yz . As with many treatment processes, achieving $E = 0.90$ is relatively easy ($S = 9$). Each additional incremental increase in E , however, becomes increasingly difficult. Thus, an S value of 99 is required for $E = 0.99$, and an S value of 999 is required for $E = 0.999$.

The fact that equation (9) and Figures 4 and 5 represent what may be expected in an actual sparging situation is supported by consideration of the well-understood gas-exchange process as discussed by Fair et al. (1973) and Gvirtzman and Gorelick (1992). These conclusions have been verified for sparging on a laboratory scale by Pankow and Johnson (1985). In that study, 1,1,1-trichloroethane and 1,1,2-trichloroethane were removed from a simulated aqui-

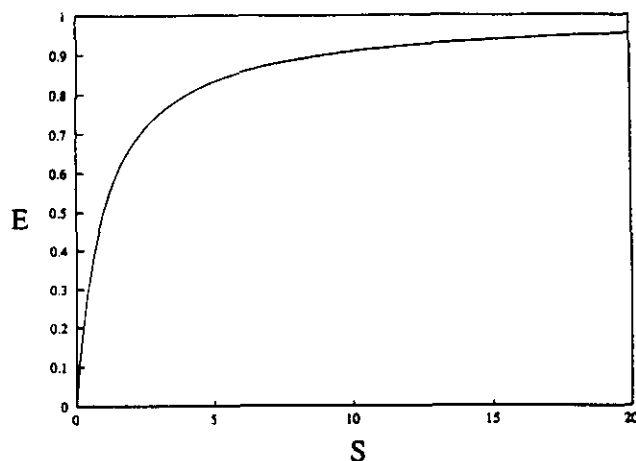


Fig. 4. Sparging efficiency E vs. S for single stage sparging with S ranging from 0 to 20.

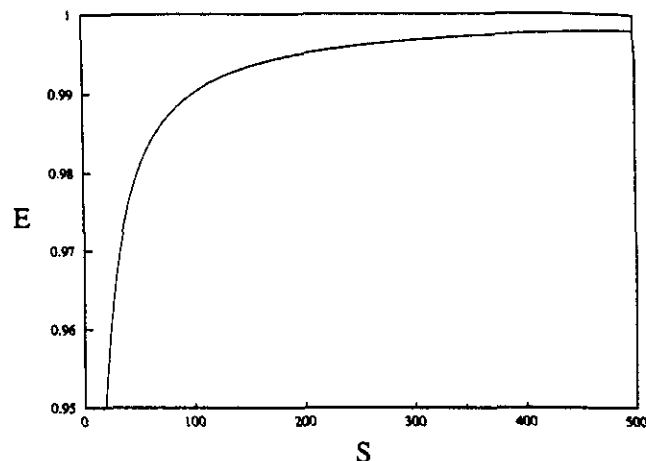


Fig. 5. Sparging efficiency E vs. S for single stage sparging with S ranging from 20 to 500.

fer system by WTS. The predicted E values for these two compounds under the experimental conditions used were 0.997 and 0.892, respectively. These values agree very well with the measured efficiencies of 0.994 and 0.946, respectively. A field sparging investigation which supports equation (9) is provided by the work of Coyle et al. (1985) using WTS for the removal of VOCs from a drinking water well while using "air lift" to pump the water to the surface.

While Figures 4 and 5 represent dimensionless plots for the determination of E for any combination of the variables comprising S , Figures 6-8 allow the examination of how E depends upon specific values of three variables. For each figure, a moderate value for one of the variables has been selected and kept constant, and the other two have been varied: one on the abscissa, and one by means of a family of curves. Since equation (10) was used to calculate S , a temperature of 15°C (288.15 K) has been assumed. Because the explicit dependence of S on T is weak, Figures 6-8 will

provide good estimates of E at other temperatures provided that H values which are correct for the temperature of interest are used, that is, provided that the stronger, implicit temperature dependence of H is taken into consideration. For Figures 6 and 7, where H is varied as a family of curves, the positions of lines for certain specific compounds at 15°C are indicated.

As seen in Figure 6, E increases with increasing R_g/yzv . However, for all $H < 0.00043 \text{ atm}\cdot\text{m}^3/\text{mol}$ (e.g., naphthalene), then $E \leq 0.90$ when $R_g/yzv \leq 500$. For a given value of yzv , a larger gas sparging rate R_g will then be needed; for $yzv = 0.0001 \text{ m}^3/\text{s}$ (e.g., $y = 10 \text{ m}$, $z = 10 \text{ m}$, and $v = 10^{-6} \text{ m/s}$ (8.6 cm/day), we would need $R_g \geq 0.05 \text{ m}^3/\text{s}$. We note that 50 hp compressors are available which will deliver 10^6 cubic feet/day ($0.33 \text{ m}^3/\text{s}$) against a pressure of 15 psig ($\sim 10 \text{ m}$ of water column head). With this R_g and with $yzv = 0.0001 \text{ m}^3/\text{s}$, the $R_g/yzv = 3300$, and even an H value of only $0.0002 \text{ atm}\cdot\text{m}^3/\text{mol}$ will yield $E > 0.95$. With terminal bubble rise velocities of $\sim 0.25 \text{ m/s}$, however, this type of R_g would not be compatible with sparging in gate wells of conventional size since a well I.D. of 1.3 m would be required to accommodate the air alone. Sparging in a trench or in a large gate well made of perforated sheet-piling could then be used. (At the water surface, the fraction f ($0 \leq f \leq 1$) of the sparging volume (of cross section $A \text{ m}^2$) occupied by bubbles can be approximated by $f = R_g/[A(0.25 \text{ m/s})]$.) In difficult cases, sparging in multiple, sequential stages to obtain efficiency multiplication may be attractive (see below).

Figure 7 presents curves of E vs. yzv for $R_g = 0.05 \text{ m}^3/\text{s}$. The lowest H values considered permit significant removal at $yzv = 0.0001 \text{ m}^3/\text{s}$, and high E values are obtained for the very volatile compounds. Figure 8 is similar to Figure 7 except that here H is held constant at a fairly low value ($0.0003 \text{ atm}\cdot\text{m}^3/\text{mol}$), and R_g is varied. As usual, high values of E are obtained for large R_g and small yzv .

The final plot of interest is one which summarizes the conditions under which a certain constant E is obtained. How large E must be in a given situation will, of course, depend upon the absolute magnitude of c_i . For $E = 0.95$,

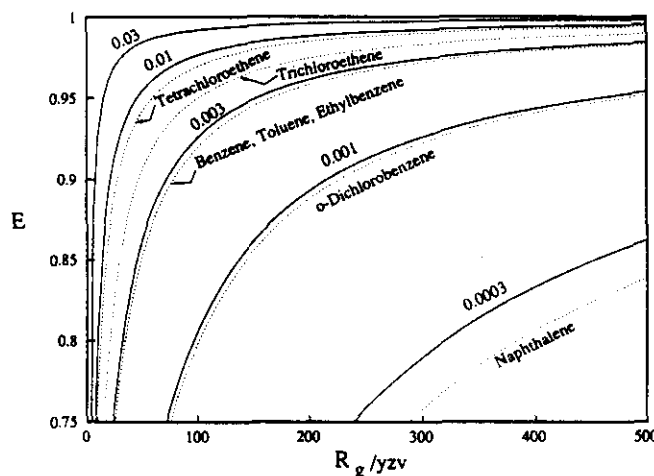


Fig. 6. Sparging efficiency E vs. R_g/yzv at $t = 15^{\circ}\text{C}$ for single stage sparging for varying H values.

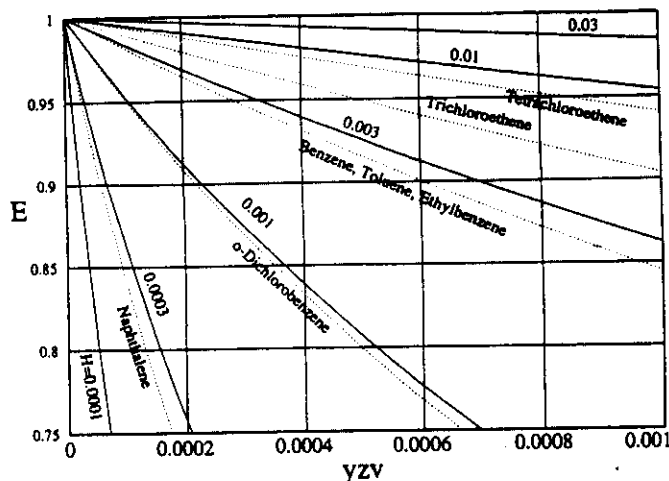


Fig. 7. Sparging efficiency E vs. yzv at $t = 15^\circ\text{C}$ for single stage sparging for varying H values, with R_g held constant at $0.05\text{ m}^3/\text{s}$.

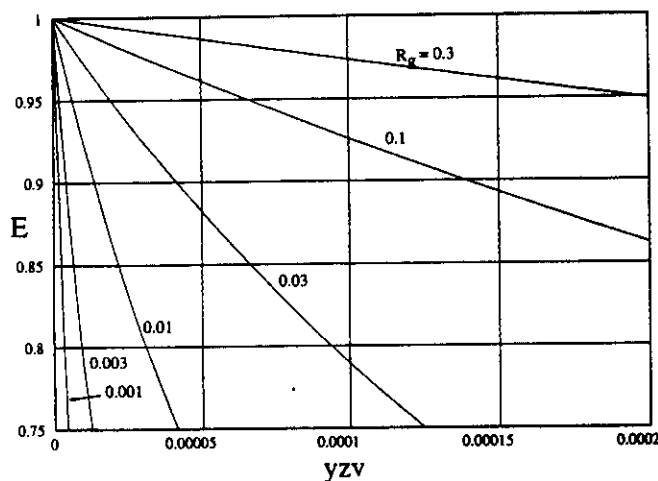


Fig. 8. Sparging efficiency E vs. yzv at $t = 15^\circ\text{C}$ for single stage sparging for varying R_g with H held constant at $0.0003\text{ atm-m}^3/\text{mol}$.

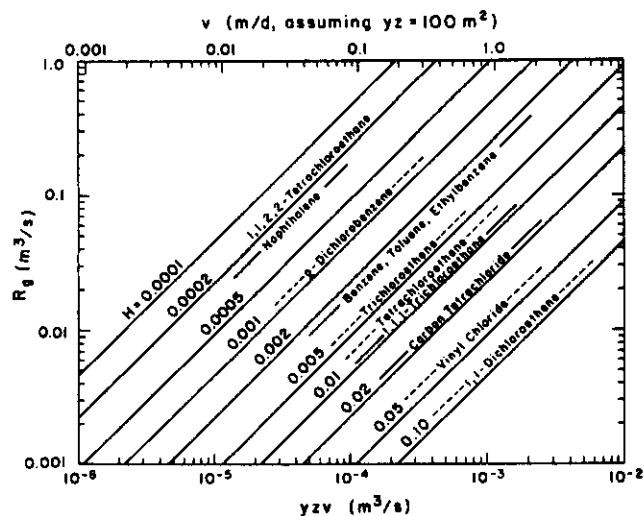


Fig. 9. R_g vs. yzv lines at $t = 15^\circ\text{C}$ which yield $E = 0.95$ for single stage sparging for varying H values. Top margin of plot labelled in terms of the Darcy velocity v in units of m/d , assuming $yz = 100\text{ m}^2$.

then by equation (9), $S = 19.0$. At $T = 15\text{ K}$, then by equation (6),

$$R_g = 0.45 yzv/H \quad (11)$$

The R_g vs. yzv lines in Figure 9 have been drawn using equation (11) for varying values of H . This plot reveals the relative ease or difficulty of applying gas well sparging under a wide range of conditions. (For purposes of comparison with specific situations, the top margin of the figure has been labelled in terms of v in units of m/d , assuming a yz value of 100 m^2 .) Thus, as yzv increases, the value of R_g required to maintain $E = 0.95$ also increases for a given H value. Also, for a given R_g and $E = 0.95$, the value of yzv that can be tolerated increases as H increases. As an example, for $E = 0.95$ and $yzv = 10^{-3}\text{ m}^3/\text{s}$ ($v = 0.36\text{ m/d}$ when $yz = 100\text{ m}^2$), if $H = 0.01\text{ atm-m}^3/\text{mol}$, we require an R_g value of $0.045\text{ m}^3/\text{s}$.

Sequential Sparging

The sparging efficiency E can be increased dramatically if the ground water is made to flow through several separate sparge cells in series rather than just through a single sparge cell. Since sparging is based on a linear gas-water partitioning process, each cell will remove contaminants with an efficiency which is independent of the contaminant concentration. The result will be a theoretical removal efficiency which is higher than that which is attainable with a single cell using the same total sparge gas flow rate.

When $S \ll 1$, then E increases approximately linearly with R_g . In the useful range of efficiencies ($E > 0.5$), we need $S > 1$. Based on Figure 4, as S approaches 1, however, E increases only slowly with R_g . Consider then a single cell sparging system in which $S = 10$ so that $E \approx 0.9$. Increasing E to 0.99 can be achieved by increasing R_g , but about 10 times the gas flow is required. However, splitting the gas flow among several different sparge cells is a much more efficient way to use the increased flow rate of sparge gas.

Consider then a series of sparge cells each of which removes a given VOC with a fractional efficiency of E . The amount remaining after passing through the first sparge cell is $(1 - E)$; the amount remaining after passing through the second sparge cell is $(1 - E)^2$. Thus, the cumulative efficiency of n sequential cells is given by:

$$E_{\text{cum}} = 1 - (1 - E)^n \quad (12)$$

We can compare the relative improvements in the overall removal efficiency obtained by using a single sparge cell vs. sequential sparge cells. For example, if in a single-cell system $E = E_{\text{cum}} = 0.50$ (i.e., S in that cell is 1), then tripling the flow in that single cell will raise E_{cum} to 0.75. However, distributing that tripled flow over three sequential cells will raise E_{cum} to 0.875, and the amount of the VOC remaining in the water is half that when $E_{\text{cum}} = 0.75$. As another example, if in a single-cell system $E = E_{\text{cum}} = 0.80$ (i.e., S in for that cell is 4), then distributing the exact same flow over two sequential cells will increase E_{cum} to 0.89 ($S = 3$ in each cell). This again reduces the amount of the VOC remaining in the water by a factor of about two. Given the low Darcy velocities present in many systems, single-cell sparging efficiencies of > 0.9

should be attainable fairly easily. Therefore, a sequence of three or four such sparge cells in either a gate well or in a trench would result in very high removal efficiencies.

Atmospheric Contamination Considerations Emissions to the Atmosphere

As is the case with aeration stripping carried out above the ground surface, sparging will produce contaminated air. If $E = 0.95$, and if c_i' is the initial concentration of the contaminant in mg/l, then the discharge in metric tons/year (t/y) will be

$$D_g = 30 yzvc_i' \quad (13)$$

It is of interest to examine how D_g for an extensively contaminated system will scale with other inputs to an urban airshed. Take $c_i' = 50 \text{ mg/l}$ ($= 50,000 \text{ } \mu\text{g/l}$) as summed over all of the volatile contaminants present. For $y = 100 \text{ m}$, $z = 20 \text{ m}$, and $v = 10^{-6} \text{ m/s}$ (8.6 cm/day), then $D_g = 3.0 \text{ t/y}$. While this may seem large in absolute terms, it is in fact not large relative to typical urban sources of volatile organic compounds (VOCs). The following are common, current emission rates: large gasoline station with vapor recycle emission controls, 2 to 3 t/y; clothing dry cleaners, 5 t/y (tetrachloroethene); large industrial "stoddard solvent" dry cleaners, 40 t/y; large industrial degreasers, 40 t/y. While the situations in individual locales vary, in general, discharge permits are often not currently necessary until the emissions rise above $\sim 10 \text{ t/y}$. Moreover, the D_g values at many ground-water contamination sites will be even lower since the total amounts of volatile contaminants present in whole ground-water systems are often less than a few metric tons. In general then, unless the political climate is relatively strict, the emissions from sparging are not likely to require regulatory attention. When the discharges are either high or inherently toxic, options that remain include: (1) acquisition of a discharge permit; or (2) treatment of the sparge air effluent prior to discharge, e.g., by sorption onto activated carbon.

Effects of Contaminants Already Present in the Urban Atmosphere

A result of normal industrial emissions of VOCs is nonzero ambient air concentrations. Thus, E as predicted by equation (9) for WTS will not be fully attainable if the compound of interest is present in the sparge air even before the air enters the sparging zone. The decrease in efficiency may be predicted based on a mass balance approach similar to that used to derive equation (9). In this case,

$$\text{mols into sparging zone/s} = c_i yz v + c_a R_g \quad (14)$$

where c_a is the ambient air concentration (mols/m³). It may then be shown that

$$E = \frac{S}{1 + S} (1 - c_a RT / H c_i) \quad (15)$$

The term $(1 - c_a RT / H c_i)$ is thus a correction factor for equation (9). It incorporates the ratio between: (1) the aqueous concentration with which c_a would be in equilib-

rium (i.e., $c_a RT / H$); and (2) c_i . For urban Portland, Oregon, Ligocki et al. (1985) reported the following gas phase concentrations in ambient air (mols/m³): trichloroethene, 1.1×10^{-8} ; tetrachloroethene, 7.3×10^{-9} ; toluene, 4.1×10^{-8} ; ethylbenzene, 1.2×10^{-8} ; and *o*-xylene, 1.2×10^{-8} . Thus, even when ground water which is only contaminated at the $1 \text{ } \mu\text{g/l}$ level is treated, E will not be reduced significantly below that given by equation (9) except under those situations when the sparge air intakes are positioned (carelessly) near a localized atmospheric contaminant source.

Two final situations are of interest. When $(1 - c_a RT / H c_i)$ is less than zero, E will be negative, and sparging will cause c_i to be greater than c_i . In a similar manner, when c_i is zero for a given compound and its c_a is nonzero, then sparging will lead to ground-water contamination where there was none before. When problematic, both of the cases are likely to occur only for compounds that are not the direct target of the sparging. However, the resultant levels of contamination will generally be extremely low, and not the subject of concern.

Conclusions

Well/trench sparging (WTS) has the potential to become a useful treatment method for removing VOCs from contaminated ground-water plumes. It is suited for use with a very large number of the solvents and petroleum products which have caused extensive ground-water contamination. The theory of the method is simple, and the theoretical removal efficiencies are predictable as well as adjustable. The advantages of the method include the facts that: (1) since the Darcy velocity v in many systems is relatively low, only a relatively small volume of water must be treated per unit time; (2) the water is not removed from the aquifer; and (3) unlike large-scale pumping and treat, it does not draw large volumes of uncontaminated water into the zone of contamination, nor does it mix large volumes of uncontaminated water with contaminated water prior to treatment. Plans are now being made to test well/trench sparging at the Borden field site.

Acknowledgment

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6375 Riverside Drive
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REMEDIATION BY IN-SITU AERATION

The Power of Volatilization and Bio-Oxidation

David B. Vance
Corporate Director of
Strategic Services
ATEC Associates, Inc.
Indianapolis, IN

INTRODUCTION

Soil venting is an in-situ aeration process that is a powerful remediation technology for the treatment of soils exposed to a variety of hydrocarbons. The practice of soil venting includes the following variations in application:

- Soil vapor extraction (SVE) systems are designed to exploit a hydrocarbons potential for volatilization.
- Air sparging is an aeration process in which volatilizing air is injected into the saturated zone beneath the water table. Soil venting is used to recover the vapor laden air as it exits the water table.
- Bioventing is an aeration process designed to deliver oxygen to the subsurface for use by indigenous bacteria to degrade hydrocarbons, the focus is on minimizing hydrocarbon volatilization.
- Biosparging is a variant of air sparging where oxygen stimulated biodegradation is the aim, rather than volatilization. As with air sparging, soil venting is used to recover gas discharged through the water table.

Figure 1 illustrates and compares the potential effectiveness of soil vapor extraction and bioventing.

The data illustrated in Figure 1 was calculated using the following premises:

- The air flow rate is 10 SCFM (Standard Cubic Feet per Minute).
- It is assumed that the 10 SCFM air stream becomes saturated with hydrocarbon vapor.
- Volatilization driving vapor pressures

were calculated at 8° centigrade.

- For biodegradation, the oxygen provided by the 10 CFM air flow is completely utilized for hydrocarbon bio-oxidation.
- Biodegradation of each of the hydrocarbons proceeded to carbon dioxide and water.

This data was derived from theoretical calculations predicated on the fundamental principals governing the action of each of the processes (i.e.

volatilization or bio-oxidation). In actuality, these processes and other subsurface interactions (with soil moisture for example) are more complex. However, the base principals do apply and are, in the overall process, upheld. Following is a more detailed explanation of these processes and a case history that illustrates the use of biosparging technology at a site contaminated with volatile and non-volatile hydrocarbons.

SOIL VAPOR EXTRACTION

Soil venting exposes adsorbed hydrocarbons to an advective air flow that does not carry an existing vapor load. Upon exposure, equilibrium driving forces will volatilize the hydrocarbons into the flowing air. Hydrocarbons have a specific temperature dependant vapor pressure determining the maximum vapor concentration that can occur. This vapor saturated air is then directed to the surface where it may be discharged to the atmosphere, or more commonly, treated to remove the hydrocarbon vapors.

The key in the vapor extraction process is that the mass transport rates are determined by a physical property (vapor pressure) of the hydrocarbon, the lower the vapor pressure, the lower the overall mass transport rate will be. Thus, the dramatic trend seen in the SVE portion of Figure 1. Table 1 presents the data used to prepare the SVE portion of Figure 1.

Benzene is extremely volatile and offers an excellent SVE mass transport potential of 166 pounds per day. However, naphthalene is at the other extreme, with a vapor pressure less than 1 mm Hg, which is at the lower range considered amenable to soil vapor extraction. In this instance only 0.13 pounds per day would be removed.

The dependance of soil vapor extraction on vapor pressure can place constraints on remediation rates. Even gasoline has significant concentrations of hydrocarbons with relatively low vapor pressures. Heavier petroleum products such as jet fuel, kerosene, diesel fuel and lubricating oil are thought to be non-responsive to soil vapor extraction technology. Based on soil vapor extraction alone, that thought is an accurate one.

Air sparging is the process of hydro-

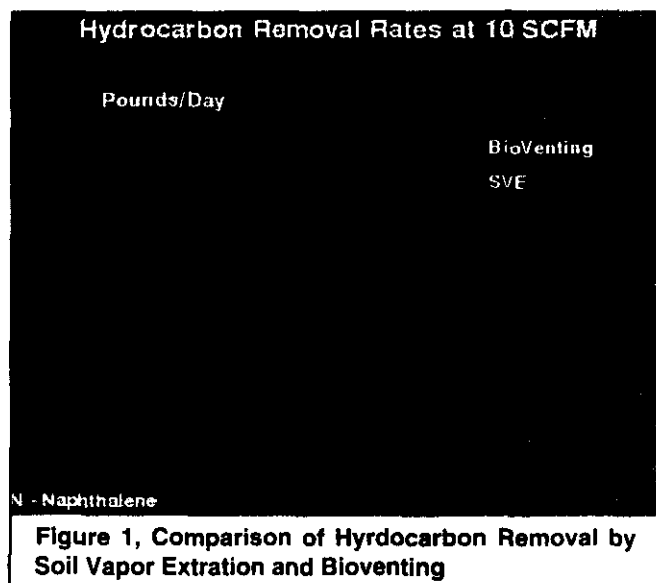


Table 1, Vapor Pressure and Volatilization Driven Mass Transport Rate

Benzene	41.0	78.1	11.5×10^{-3} lbs/ft ³	166
Ethylbenzene	5.5	106.2	2.10×10^{-3} lbs/ft ³	30.2
Naphthalene	0.02	128.2	9.2×10^{-6} lbs/ft ³	0.13
* mm Hg at 80 Centigrade				

Air sparging is the process of hydrocarbon volatilization by injection of air into the water table. The volatilization process takes place under saturated conditions, emulating the action of an air stripping surface treatment system. The governing physical parameter that relates a hydrocarbons volatilization potential from water is the Henry's constant. Which is directly related to vapor pressure, water solubility and temperature (Haarhoff and Cleasby, 1990).

In an air sparging system the injected air and entrained volatilized hydrocarbons are captured above the water table with a conventional soil venting system.

BIOVENTING

Bioventing is the term for aerobic biodegradation stimulated by oxygen introduced through soil venting. The power of bioventing lies in the stoichiometry of the oxidation of hydrocarbons to carbon dioxide and water.

The data calculated for the bioventing portion of Figure 1 is based on the stoichiometric consumption of oxygen. The exact stoichiometry is as follows:

Table 2, Bio-Oxidation Stoichiometry

Benzene	C_6H_6	+	$7.5 O_2$	\rightarrow	$6 CO_2 + 3 H_2O$
Ethylbenzene	$C_6H_5(C_2H_5)$	+	$10.5 O_2$	\rightarrow	$8 CO_2 + 5 H_2O$
Naphthalene	$C_{10}H_8$	+	$12 O_2$	\rightarrow	$10 CO_2 + 4 H_2O$

A flow rate of 10 SCFM can deliver 270 pounds of oxygen to a treatment zone in a period of 24 hours. Based on the above, the potential biodegradation rates in pounds per day are as follows:

Table 3, Bio-Oxidation Rate

Benzene	88
Ethylbenzene	86
Naphthalene	90

This entire process is independent of any other physical property of these hydrocarbons. The prime issue with regards to bioventing is effective oxygen transport. However, there are some limits to the effectiveness of biodegradation. The problems are with compounds that are recalcitrant to biodegradation. With petroleum hydrocarbons, these recalcitrant compounds are typically polynuclear aromatic (PNA) compounds having high ring counts. However, in many products high ring count PNAs are not a significant amount

of the total hydrocarbon makeup. In addition, these compounds are most often still biodegradable, but at a slower rate.

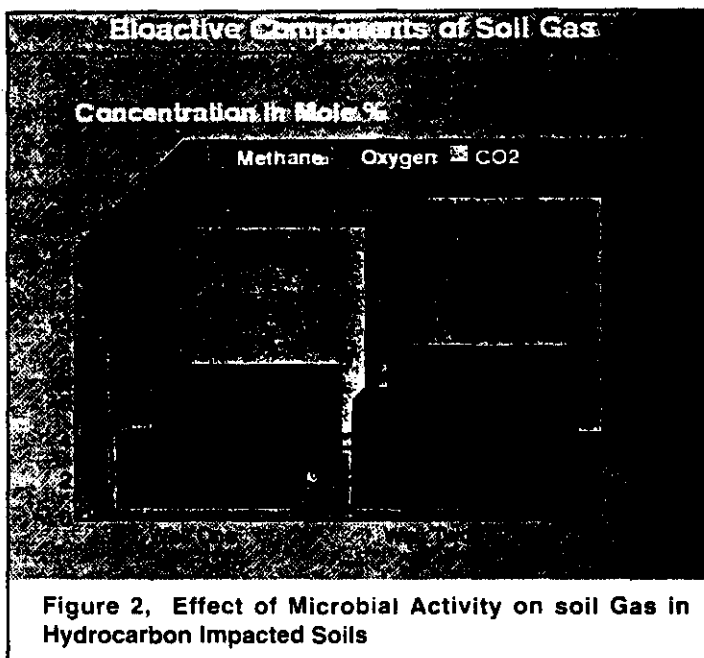
Indigenous Bacteria

A concern for the viability of bioventing is the presence of indigenous bacteria capable of being stimulated to degrade hydrocarbons.

Figure 2 illustrates soil gas data from a site impacted with petroleum hydrocarbons. These samples were collected under static conditions, no remediation activity has taken place.

Under normal atmospheric conditions oxygen concentration is 21 percent.

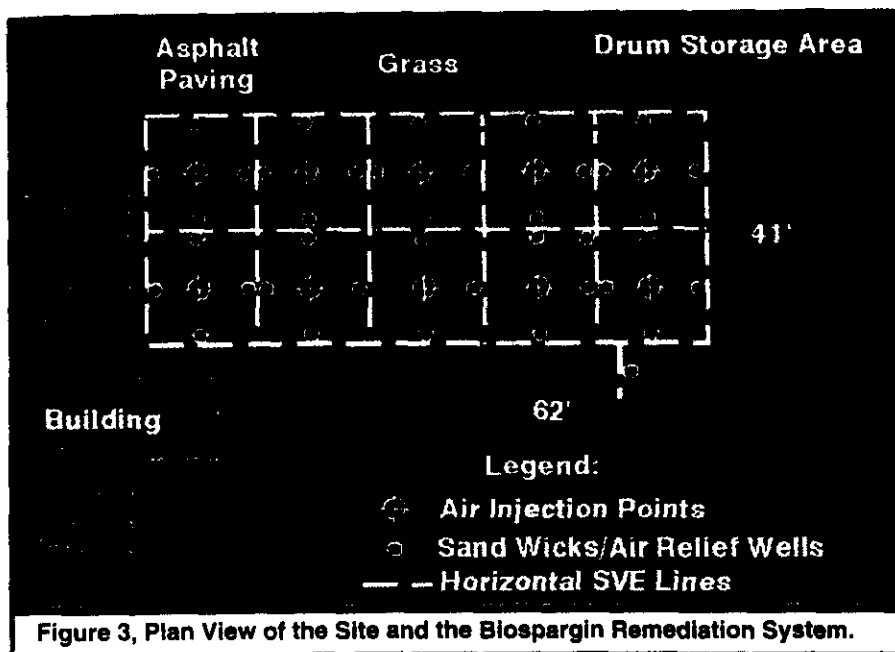
Carbon dioxide is approximately 300 ppm (0.03%). Due to the presence of carbonate minerals and natural organic materials, CO_2 concentrations in uncontaminated soils are in the range of 1.5 to 3% (Suchomel et al, 1990). Methane is present in the atmosphere in trace amounts (1.5 ppm). As Figure 2 illustrates, the soil gas concentrations at this contaminated site are significantly skewed from those levels. Carbon diox-



ide is elevated at 11% to 12%, oxygen is depressed to around 5% and methane is elevated at 3% to 5%.

Following is an outline of the process responsible for the generation of this soil gas blend:

- Upon release of hydrocarbons into the subsurface the indigenous bacteria began aerobic bio-oxidization. The end product of this aerobic microbial degradation was carbon dioxide and water (see Table 2).
- After the aerobic microbial activity had consumed oxygen in the soil gas to near the observed 5% level, facultative anaerobes became active. These bacteria have the ability to support metabolic activity under full aerobic or oxygen depressed conditions. This anaerobic degradation occurs at a rate several orders of magnitude slower than that observed for aerobic degradation (Atlas, R.M., 1981). If this were not the case, it would be cost effective to let the anaerobic degradation occur at its own pace with no other intervention.
- The degradation products (seen in the gas phase) of the anaerobic activity are methane and additional carbon dioxide. The fundamentally important point of



the data illustrated in Figure 2 is that this aerobic/anaerobic activity occurred naturally. There were no bacteria added to the soil, the existing indigenous bacteria generated these gases and there were no nutrients added.

The processes engaged in biosparging are identical to those in bioventing. The biosparging stimulated bio-oxidation follows the same stoichiometry presented in Table 2. The injected air, bio-oxidation products (CO_2) and some fraction of volatilized hydrocarbons are collected above the water table with a soil venting system, just as described for air sparging.

SOIL VAPOR EXTRACTION VERSUS BIOVENTING - A SIMULTANEOUS PROCESS

A soil venting system operated at an impacted site will engage both the discussed mechanisms. The determination of which process is dominant lies in how the system is operated.

A soil venting system can be operated such that 60 to 90 percent of the hydrocarbons are volatilized and 10 to 40 percent are biodegraded, i.e. soil vapor extraction. Bioventing reverses those numbers, 60 to 90 percent of the hydrocarbons are biodegraded and 10 to 40 percent are volatilized (Miller et al, 1990).

Soil vapor extraction is definitely the preferred approach when remediating biological recalcitrant compounds that have high vapor pressures such as chlorinated solvents.

However, in instances where the released materials are petroleum hydrocarbons such as fuels or lubricants, bioventing is likely to be the most cost

and time effective remediation approach. Even if equal times are required, the effective flow rates for bioventing are much lower than for SVE.

BIOSPARGING - A CASE HISTORY

Following is a field example of the biosparging process discussed above.

This project is on-going at a facility which was closing a RCRA hazardous waste drum storage area (DSA). The wastes stored in the DSA were "Spent Non-Halogenated Solvents" (F003/F005). Figure 3 shows a plan view of the site, the DSA, and the installed remediation system.

The geology beneath the site consists

of approximately 30 feet of sand with a peat layer 1 to 3 feet thick at a depth of 16 feet. Silt and clay underlies the sand. Depth to groundwater is 6 feet (see Figure 4). Soil and groundwater contamination was limited to the sand/peat unit above the clay. Soil contamination included: ethylbenzene; toluene; total xylenes; naphthalene, all in the 100 to 600 mg/kg range and other polycyclic aromatics at low mg/Kg levels. Groundwater contained 2 mg/L ethylbenzene and 16 mg/L total xylenes.

The remediation system was an in-situ saturated zone treatment using biosparging. The details (Figure 4) of which are as follows:

- A series of sparge points were installed to a depth of 30 feet, just above the surface of the lower clay layer.
- A series of 4 inch borings, filled with graded sand to act as air relief wells (sand wicks), were installed to a depth of 30 feet. The sand wicks act as a conduit for injected air bubbles through the peat layer, preventing unacceptable horizontal migration of the injected air and any entrained vapor. This is a common and critical problem for the application of air sparging/biosparging in heterogeneous soils.
- A soil venting recovery system was installed in horizontal trenches above the water table to capture the injected air and hydrocarbon vapors generated from the biosparging system. The area was then covered with a plastic liner and soil to prevent short circuiting.

Figure 5 is a photograph of the site that shows the vapor phase activated carbon system, air compressor (for injection air) and soil vapor recovery pump.

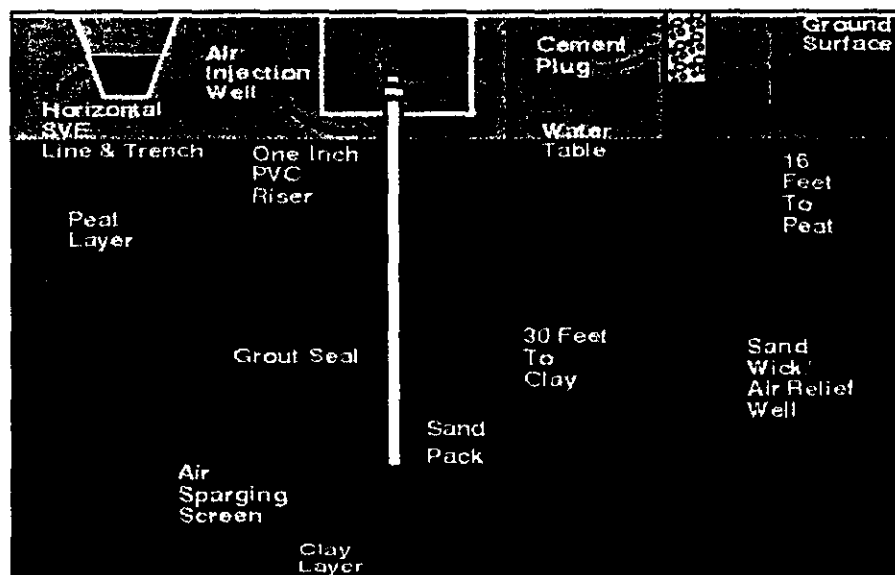


Figure 4, Cross Sectional View of Biosparging System



Figure 5, View of Site and Biosparging Equipment Area

The intent of the remedial design was to minimize the actual volatilization and concentrate on oxygen stimulated biodegradation of the hydrocarbons in-situ. To this end, air injection was only at 12 SCFM, a rate estimated to match the kinetics of the microbiological bio-oxidation. The horizontal soil venting system was operated at 60 SCFM to insure that all the injected air was recovered.

While provisions were made for the addition of nutrients (nitrogen, phosphorus and trace minerals), nutrients were not actually utilized in this phase of the project.

Figures 6 illustrate the results after system start-up showing the trend of VOCs and CO₂ observed in recovered soil gas. This trend is dominated by an increase in the concentration of VOCs recovered by day 10 of operation. Initial carbon dioxide levels are quite low. In a manner very typical of microbiological systems, carbon dioxide concentrations steadily increased to over 30 days. This period is termed the "lag phase", as the indigenous microorganisms adapt to the introduction of oxygen into their environment and utilization of the hydrocarbons as a carbon source.

In turn, the VOC concentrations continue a steady decline. The most striking feature of this data is the continuous generation of significant amounts of carbon dioxide after day 30, with very low concurrent VOC emissions. This is due primarily to the bio-oxidation of nonvolatile hydrocarbon components (such as lighter PNAs), which are not responsive to volatilization.

After 50 days of operation the CO₂ levels precipitously decline, in conjunction with VOCs becoming almost undetectable. This indicates that the hydrocarbons have been consumed.

This site data has been presented to provide a topical field example of biosparging. It is from the early stages of the remediation. The hydrocarbon attenuation indicated by Figure 6 is occurring in the advective zone of the impacted saturated zone soils and groundwater. What still must be addressed are adsorbed hydrocarbons associated with low permeability diffusional transport zones and with the peat layer (Vance,

D., 1993 a and b). This portion of the site remediation is still in progress.

In conclusion, aeration is a powerful remediation tool in the vadose and saturated zones. The manner in which an aeration system is operated will determine the dominant process stimulated, volatilization or bio-oxidation. Bio-oxidation has distinct advantages with regards to the range of non-volatile hydrocarbons that can be remediated and offers lower potential operating and off-gas treatment costs. The core issue is to understand the underlying mechanisms responsible for both processes and utilize that knowledge accordingly.

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Biosparging - VOC and CO₂ Concentrations in Soil Gas

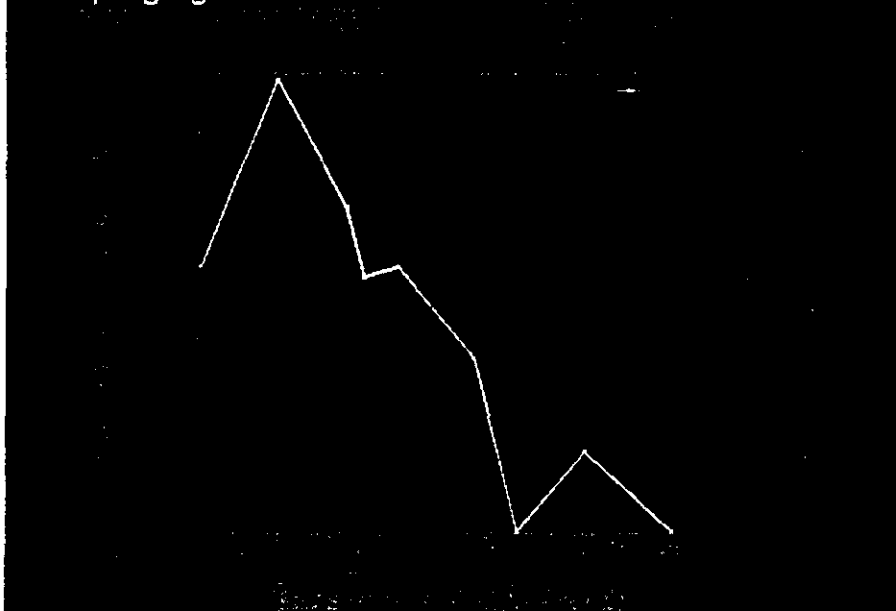


Figure 6, VOC and CO₂ Concentrations in soil Gas Collection from a Biosparging System

APR 24 1995

A.S.T.M.

Paul D. Lundegard¹, and Greg Andersen²

NUMERICAL SIMULATION OF AN AIR SPARGING FIELD TEST

REFERENCE: Lundegard, P. D., and Andersen, G., "Numerical Simulation of an Air Sparging Field Test," Subsurface Fluid-Flow (Ground-Water and Vadose Zone) Modeling, ASTM STP 1288, Joseph D. Ritchey and James O. Rumbaugh, Eds., American Society for Testing and Materials, Philadelphia, 1996.

Abstract: The flow behavior of an air sparging pilot test was simulated using a finite difference, multiphase flow simulator (TETRAD). The field test is one of the only examples where the airflow pattern in the saturated zone is well known. This is a result of the relative homogeneity of the aquifer and the use of an advanced geophysical monitoring technique known as electrical resistance tomography (ERT). ERT is sensitive to the changing water content of the saturated zone during the test and provided a clear image of the size and shape of the principal region of airflow. In addition to ERT results, slug tests and core analyses provided other key calibration data. The multiphase flow simulations provided a good match to the observed pattern of airflow and pressure changes, indicating that such simulations may be useful for evaluating air sparging performance under other conditions.

KEYWORDS: air sparging, simulation, modeling, multiphase flow

Air sparging is a method for remediating ground water and soil by injection of air into the saturated zone of an aquifer via a well or wells [1]. The goal of air sparging is to reduce the mass of dissolved and sorbed contaminants by removing volatile constituents in the air stream and accelerating aerobic biodegradation by oxygenating the subsurface. Since both these processes involve mass transfer between air and water, the subsurface pattern of airflow is an important factor in determining optimal placement of injection wells and overall remedial efficiency. Field pilot tests are often used to obtain site specific information about the behavior of an air sparging well(s) at a given site. The flow behavior of air injected into the saturated zone is difficult to define in the field [2] and has been shown to depend on a variety of geological and engineering factors [3]. In this paper a detailed air sparging pilot test is simulated. The pilot test was monitored using an advanced geophysical technique known as electrical resistance tomography, or ERT. ERT provided a very good description of the distribution of injected air in the saturated zone which could be used for model calibration.

¹ Research Associate, Environmental Technology Group, Unocal Corp., Brea, CA, 92621

² Research Associate, Production Technology Group, Unocal Corp., Brea, CA, 92621

FIELD TEST

The air sparging pilot test that is simulated in this paper was conducted in a shallow, dune sand aquifer. The test was the focus of a detailed investigation designed to determine the flow response in both the saturated zone and unsaturated zone, and to evaluate the utility of conventional monitoring measurements for estimating the region of airflow in the saturated zone [2]. A brief summary of the site conditions and test response is given here.

below

The site investigated in this study is in the western Oregon town of Florence at an elevation of 9 m above sea level. It is a former service station at which soil and ground water in an unconfined aquifer were contaminated by petroleum hydrocarbons, principally gasoline. At the time of the air sparging test, the property was predominantly unpaved and surrounded by paved streets, parking lots, and sidewalks. Native soil and fill were exposed at the surface of the property. The air sparge well was installed in a 20 cm (8 inch) diameter hole advanced by a cable tool rig and was constructed of 5 cm (2 inch) diameter PVC pipe with a 0.6 m long V-wire screen section consisting of 0.5 mm-wide openings. The top of the well screen was approximately 4.3 m below the water table at the time of the test.

Subsurface sediments at the site are very uniform and consist of sand and gravel fill to a depth of 0.76 m, below which there is Quaternary eolian dune sand. No low permeability zones such as beds of silt or clay were encountered in any of the drilling at the site. Texturally the sand is very well sorted and medium grained. Mean grain size is approximately 0.25 mm. At the time of the field testing the water table was approximately 5.2 m below land surface and had a gradient of approximately 0.006 m/m towards the southeast. Slug tests of the sparge well indicated a horizontal conductivity of between 5×10^{-3} and 4×10^{-2} cm/s (Table 1). Other investigations of the hydraulic properties of the aquifer in the vicinity of Florence reported conductivities ranging from 1.2×10^{-2} to 2.8×10^{-2} cm/s. Measurements made on core samples, collected with a 6.35 cm (2.5 inch) diameter split tube sampler, indicated a slightly lower average horizontal conductivity than the slug tests, possibly due to compaction induced during sampling. The ratio of horizontal and vertical permeabilities in adjacent core plugs ranged from 2.5:1 to 0.85:1. It is likely that such data on small core plugs provide a minimum estimate of the true anisotropy of the aquifer. Porosity measurements averaged 40 %, with little variation (Table 1). Air-water capillary pressure curves under drainage conditions were determined for two core samples. These samples showed residual water saturations in the range of 11 to 17%, very low air entry pressures (< 3.5 kPa), and steep capillary pressure curves.

Table 1 - Summary of Data on Aquifer Physical Properties

Property	Value
Porosity	0.40 +/- 2.6
Core Permeability-horizontal	13.2 +/- 6.4 darcys
Core Permeability-vertical	8.8 +/- 2.8 darcys
Hydraulic Conductivity (slug test)	$0.5-4 \times 10^{-2}$ cm/s

Prior to the start of the pilot test the air compressor was adjusted to a maximum operating pressure that was safely below the fracture

pressure of the aquifer. Injection rate was not constrained by the compressor so that the actual flow rate was determined by the subsurface aquifer conditions alone. This approach is believed to provide improved remedial performance by maximizing the region of airflow, and also provides additional information for calibration of the flow model.

As is typical of the initial transient behavior of air sparging systems, the injection pressure rises rapidly to a peak once the air compressor is turned on, and then gradually declines to a steady state value (Fig. 1). At the same time, injection flow rate gradually rises to a steady state value (Fig. 1). At steady state, the injection pressure was 41.4 kPa and the flow rate varied from 0.42 to 0.48 m³/minute.

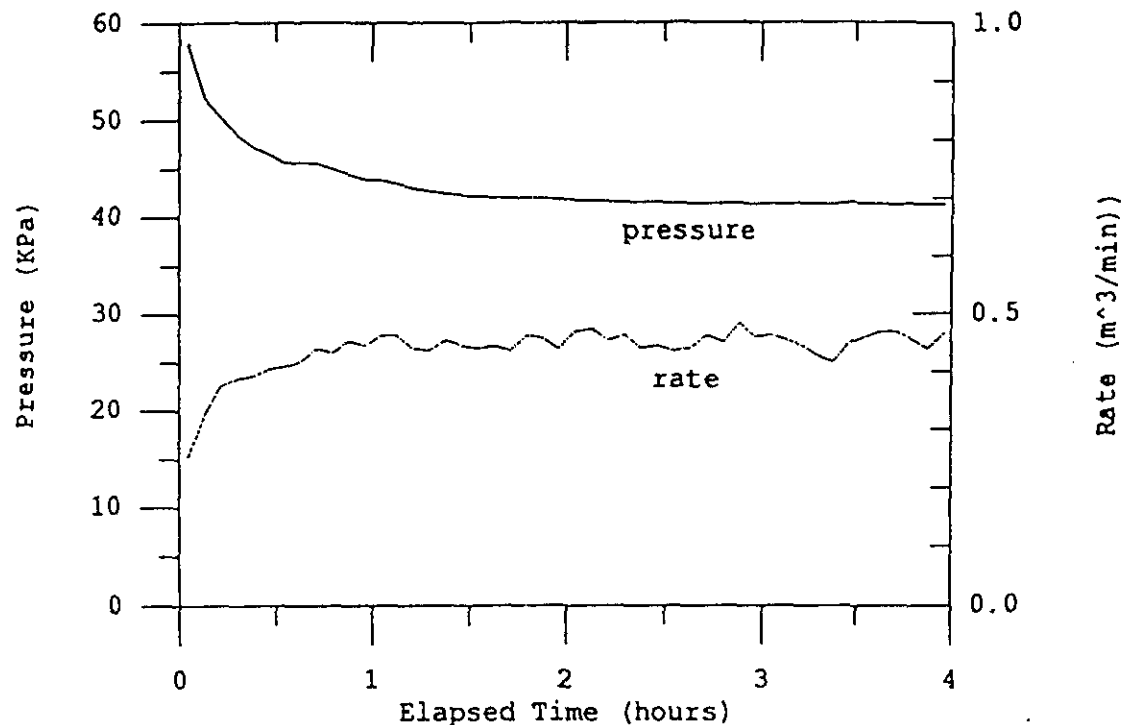


Figure 1 - Pilot test injection pressure and rate versus time.

During the pilot test vertical displacement of the water table (mounding) was monitored. The transient mounding response in three wells, each 4.57 m from the sparge well, was very similar (Fig. 2). this behavior is evidence that the aquifer is relatively homogeneous, a trait that was used to simplify the simulations.

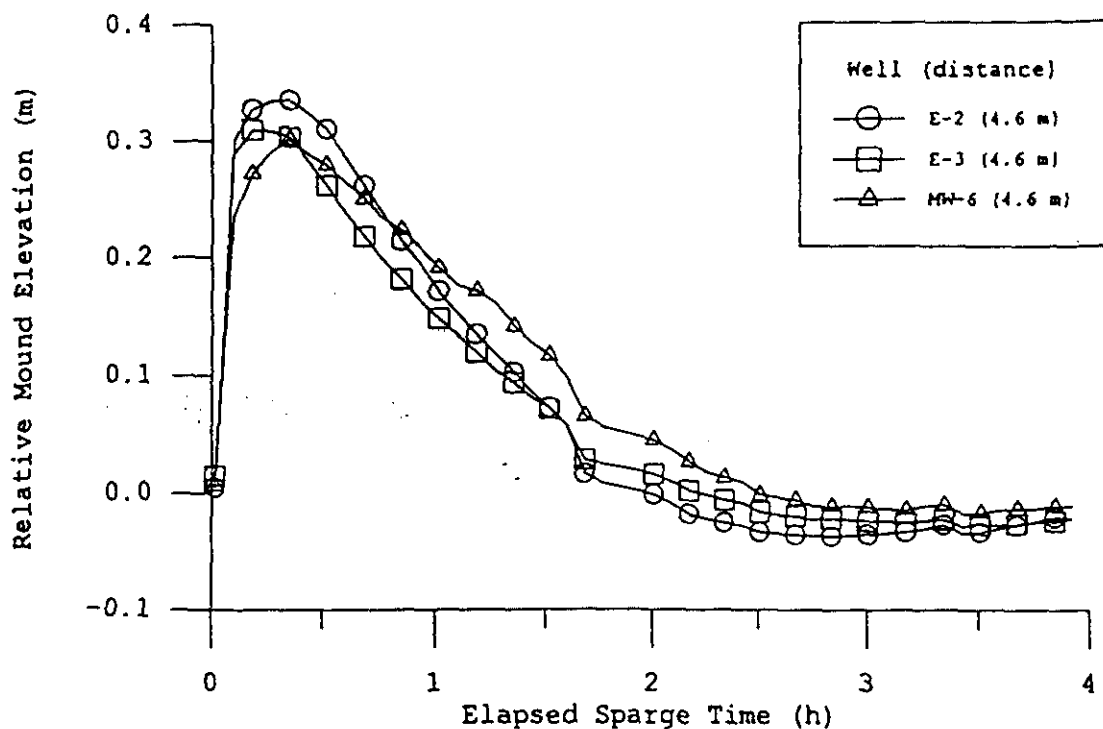


Figure 2 - Water table mounding versus time for three well equidistant from the sparge well. Note close approximation to radial symmetry.

A special part of the pilot test was the use of an innovative geophysical technique known as electrical resistance tomography, or ERT. ERT measures changes in the bulk electrical resistance of the subsurface and is very useful for in situ monitoring of air sparging because electrical resistance is very sensitive to changing water content [4]. Resistance changes were monitored between the sparge well and two ERT monitoring wells, each 4.6 m away from the sparge well and in orthogonal directions. The ERT results defined a principal region of airflow that, at steady state, was approximately 2.4 m in maximum width (Fig. 3), and differed only slightly in the two directions. Results are presented here as contours of percent change in resistivity relative to pre-sparging background values across the vertical plane between the sparge well and one ERT well. The 100% change contour coincides approximately with an air saturation of 0.2. The region of airflow widens slightly between the point of injection and the water table. Some air moved downward slightly from the point of injection. The gross pattern of airflow imaged by ERT at this geologically simple field site is quite similar to what has been observed in the laboratory with artificial sand [5].

The ERT images have a resolution of approximately 0.5 m. Consequently, they provide a very good representation of the megascopic pattern of air distribution. They do not, on the other hand, allow identification of individual air channels of the size (few cm's in diameter) that some workers have suggested might characterize the flow of air in the saturated zone [1].

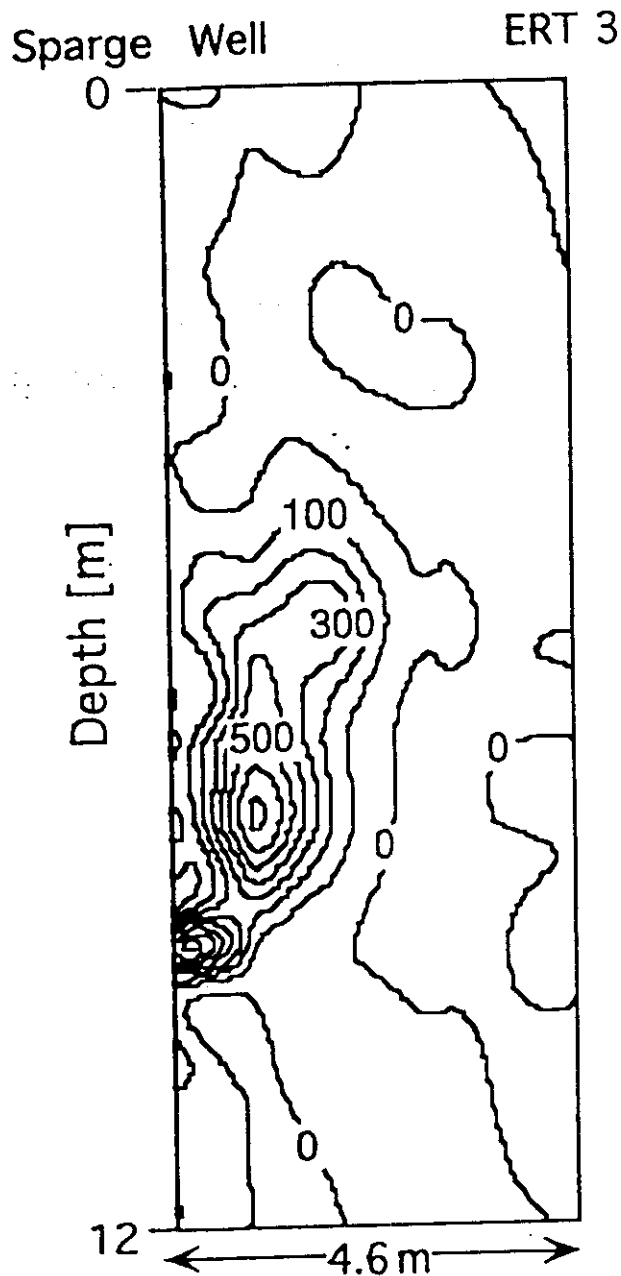


Figure 3 - ERT cross sectional image of percent change in resistivity between the sparge well and well ERT-3, 4.6 m away. Image represents conditions when the system was near steady state and defines a principal region of airflow less than 2.4 m in width.

SIMULATIONS

Air sparging is a multiphase flow process, involving the simultaneous movement of air, water, and possibly liquid hydrocarbons. The simulations presented here employ standard multiphase flow theory applicable to the representation of megascopic flow phenomena. Darcy's

law for single phase flow is modified so that the permeability term is fluid-specific. The effective permeability of the media to a specific fluid is expressed as the product of the media's intrinsic permeability and a dimensionless relative permeability which varies from 0 to 1. The modified form of Darcy's law becomes

$$q = -kk_{ri}(\rho_i g / \mu_i) (\Delta h / \Delta l)_i \quad (1)$$

where

q = the volumetric flow rate per unit area

k = the intrinsic permeability

k_{ri} = the relative permeability to fluid i

μ_i = the viscosity of fluid i

ρ_i = the density of fluid i

$(\Delta h / \Delta l)_i$ = the gradient of head.

When two fluids, such as air and water, are present in the pores of a material, the relative permeability to each fluid depends on the volumetric fraction of the pore space that it fills. This fraction is known as the fluid saturation. Furthermore, there generally is a minimum saturation that must be exceeded before a fluid becomes mobile ($k_{ri} > 0$). This minimum saturation is known as the residual saturation for that fluid. It is common for the residual saturation of the wetting phase (normally water), to exceed the residual saturation of the non-wetting phase (air in the case of air sparging). The exact relationship between relative permeability and fluid saturation depends on the properties of the fluids involved, and the properties of the porous medium.

The simulations were performed with a multiphase, multicomponent simulator known as TETRAD (DYAD 88 Software Inc.). TETRAD is a finite difference simulator, originally developed for the study of multiphase fluid flow and heat flow problems encountered during exploitation of petroleum and geothermal resources. It has been modified for the purpose of the present simulations to allow a constant pressure surface boundary that permits entry or exit of various fluid phases. The formulation structure and solution methods of TETRAD have recently been described in detail [5].

Two fluid phases are included in the present simulations, water and air. Isothermal compressibility of both fluid phases is considered, but is only significant for the transport of the gas phase.

Grid and Boundary Conditions

TETRAD is formulated to simulate multiphase flow in three dimensions and in complex, heterogeneous, anisotropic systems. Because of the radially symmetric response of water table mounding during the test and the similarity of the ERT response in orthogonal directions, a radially symmetric grid with a single air injection well at the center was selected for the simulations (Fig. 4). In this mode, flow calculations are done in a radial coordinate system, allowing for radial changes in block volumes and interblock areas. Block lengths in the circumferential direction, are $2\pi r$, where r is the radius to the center of the grid block. Grid blocks were 30.5 cm (1 ft) in radius for the first 6.1 m (20 ft), beyond which they were progressively increased by a factor of 1.5. In the vertical direction a grid spacing of 30.5 cm (1.0 ft) was used, except in the vicinity of the water table where a spacing of 7.625 cm

(0.25 ft) was used to more precisely represent the behavior of ground water mounding. The final radial grid consisted of 29 columns and 54 rows, representing a total depth of 12.2 m (40 ft) and total radius of 48.5 m (159 feet) (Fig. 4). The grid radius was large enough that lateral boundary effects were negligible.

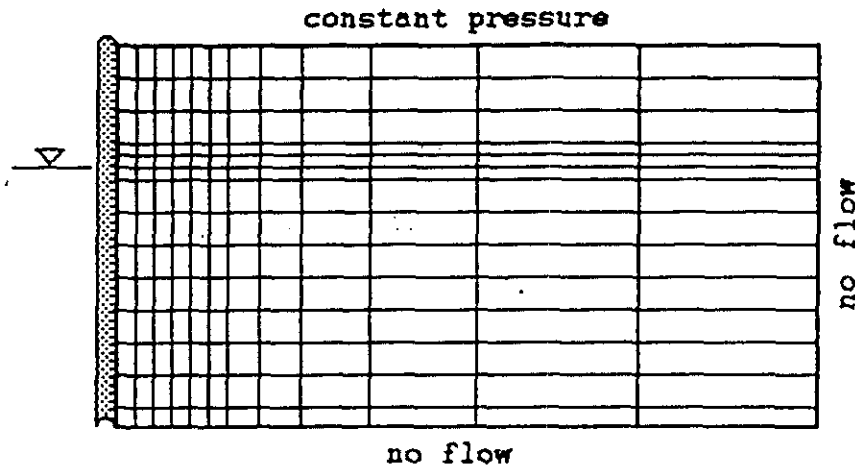


Figure 4 - Schematic radial cross section of the grid used for the simulations.

Description of Well Model

In TETRAD, recovery and injection wells can be completed in either the X, Y, or Z directions, and can have multiple completion intervals (i.e. more than one grid block in which the well is completed. Most ground water modeling codes treat wells as simple line sources or sinks with a specified flow rate. In TETRAD, well performance is determined by two factors, the well productivity index and well constraints. Well constraints consist of pressure and rate limits. For an injection well a maximum bottom hole pressure is specified, as well as a maximum injection rate for each component being injected. Either the pressure constraint or the rate constraint will limit well performance at a given time. When a well is first turned on the pressure constraint will first limit the well performance until the rate constraint is reached. The well productivity index (PI) determines the flow rate as a function of the pressure drop between the well bore and the grid block, for a given phase relative permeability and viscosity. Pressure drop is estimated by the method described by Peaceman (7). The productivity index for each completion interval of a vertical well is calculated as follows:

$$PI = \frac{C \cdot 2.0 \cdot \pi \cdot \text{DELZ} \cdot \sqrt{k_x k_y}}{\ln \left(\frac{GF \cdot \sqrt{\text{DELX} \cdot \text{DELY}}}{RW} \right)} \quad (2)$$

where

C = a constant for dimensional uniformity
 DEL(X,Y,Z) = grid block dimensions

k_x, k_y = permeabilities in the x and y directions
 GF = a geometrical factor accounting for grid block geometry, dimensions, and boundary conditions
 RW = wellbore radius

The downhole flow rate of any phase (Q_i) is then calculated by:

$$Q_i = (k_{ri} / \mu_i) * PI * (P - PBH) \quad (3)$$

where

k_{ri} = phase relative permeability
 μ_i = phase viscosity
 PI = productivity index
 P = grid block pressure
 PBH = wellbore pressure

Model Input

Aquifer properties representative of the core data and slug tests discussed earlier were assigned to the grid for initial simulations. Homogeneous, anisotropic conditions were assumed. Simulations used relative permeability curves similar to ones published for gas-water flow in similar type sands ([6] (Table 3), and standard water and air properties (Table 2). Isothermal compressibility of gas according to the ideal gas law was assumed for air. Water was assumed to be incompressible. Pressures and saturations throughout the grid were initialized by specifying the depth of the water table (where air-water capillary pressures equal zero) and balancing the gravity and capillary pressures.

Table 2 - Relative permeability and capillary pressure values

S_v	k_{rw}	k_{rg}	P_{cgv} (kPa)
0.00	0.00	1.000	
0.05	0.000	0.897	
0.10	0.000	0.800	
0.15	0.000	0.709	10.07
0.20	0.003	0.623	6.73
0.25	0.014	0.543	5.45
0.30	0.031	0.468	4.70
0.35	0.055	0.399	4.19
0.40	0.086	0.335	3.79
0.45	0.125	0.277	3.47
0.50	0.169	0.224	3.20
0.55	0.221	0.177	2.96
0.60	0.280	0.136	2.74
0.65	0.346	0.100	2.54
0.70	0.419	0.069	2.35
0.75	0.498	0.044	2.16
0.80	0.585	0.025	1.97
0.85	0.678	0.011	1.76
0.90	0.778	0.003	1.52
0.95	0.885	0.000	1.21
1.00	1.000	0.000	0.00

Table 3 - Physical parameter values used in simulation

Parameter	Value
Water Density	1000 kg/m ³
Water Viscosity	0.001 Pa.s
Gas Viscosity	0.000018 Pa.s

Calibration

The primary focus of the calibration runs was to reproduce the steady state air saturation pattern imaged by ERT (Fig. 3), and the observed injection pressure and rate (Fig. 1), while honoring the measured range of aquifer properties. This was accomplished by setting the injection rate and then adjusting aquifer properties until the air distribution pattern best matched the ERT results, and the calculated injection pressure was in good agreement with the observed injection pressure. During the simulations, injection pressure is not allowed to exceed the maximum pressure which the air compressor was capable of producing (80.7 kPa).

A satisfactory match to the injection pressure history (Fig. 5) and to the steady state pattern of air distribution (Fig. 6) was obtained with a porosity of 0.40, a horizontal permeability of 30 darcy's, and a vertical permeability of 15 darcy's. These aquifer properties are representative of those actually measured at the site (Table 1). Results were found to be most sensitive to intrinsic permeability values. Vertical permeability had a stronger impact on the steady state results. Horizontal permeability had more of an effect on the early transient flow behavior than the behavior at steady state.

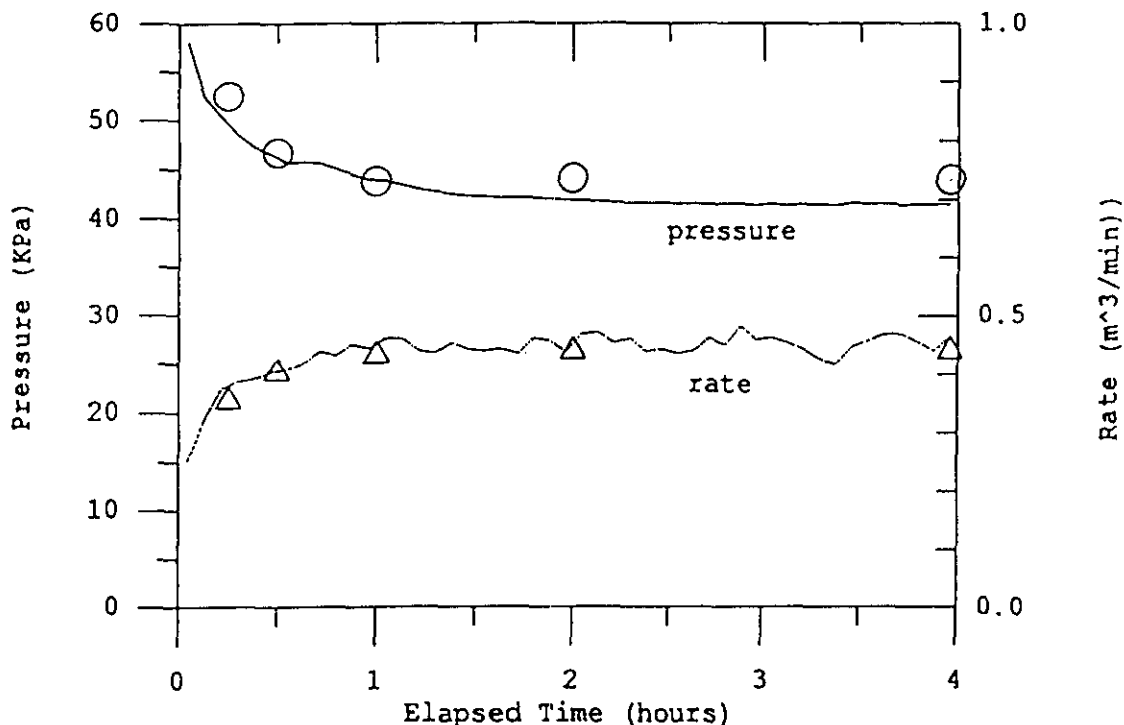


Figure 5 - Comparison of observed injection pressure and rate (solid and dashed lines) with model injection pressure and rate (circles and triangles).

Sparge Well

ERT 3

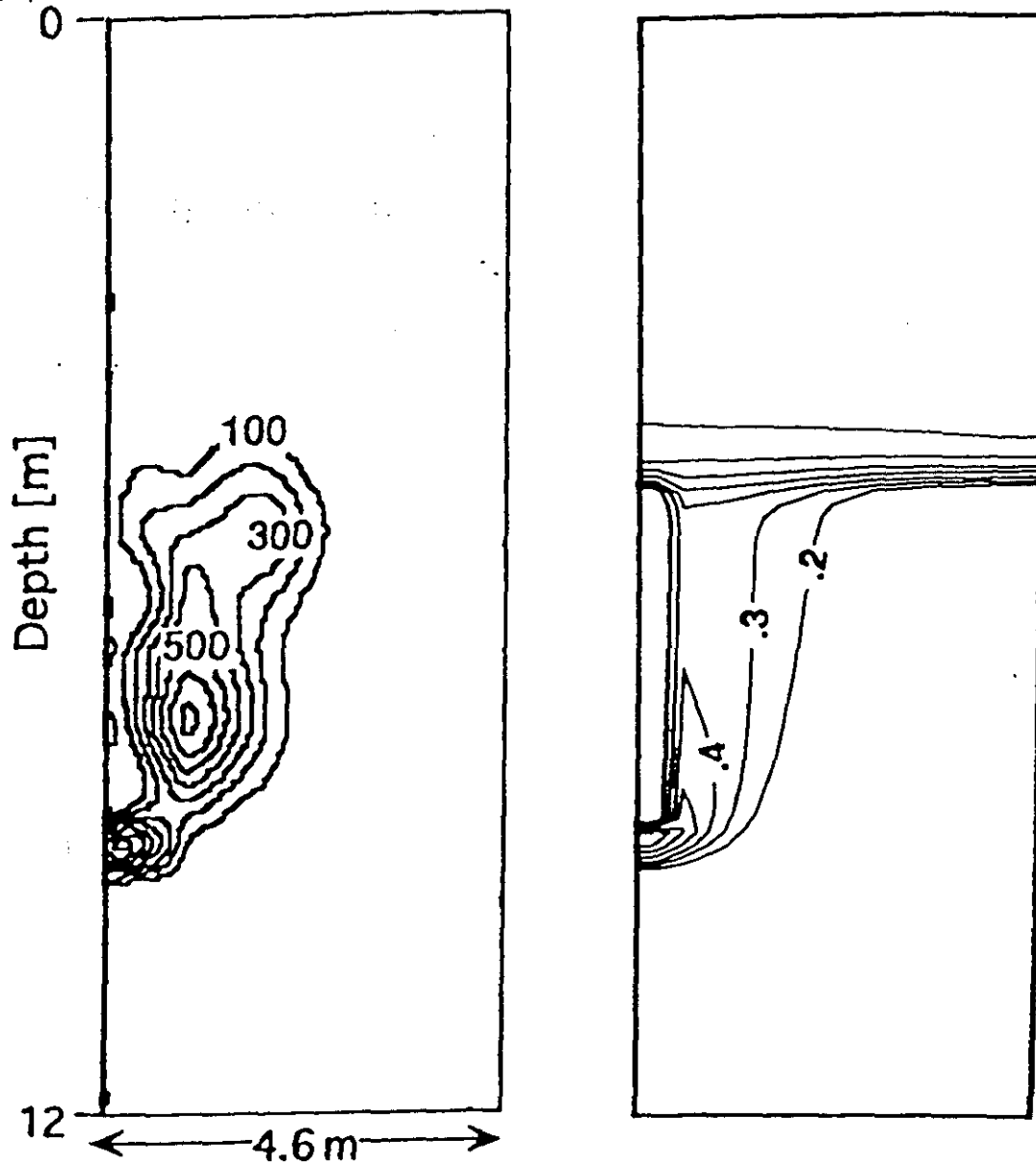


Figure 6 - Comparison of ERT image of air distribution (left) (contours in percent change in resistivity) and the simulated air saturation pattern (right). Both panels represent conditions that are close to steady state. The 100% contour on the ERT image approximates an air saturation of 0.2.

CONCLUSIONS

The behavior of an air sparging pilot test in a dune sand aquifer was successfully simulated with a multiphase flow model. The transient injection pressure history and the steady state air distribution in the saturated zone were well matched by a model which used the observed injection flow rate history and aquifer properties as input. The successful results of these simulations suggest that similar type simulations should be of value to understanding of air sparging performance and system design.

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